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US Army Corps of Engineers

WATER OPERATIONS
TECHNICAL SUPPORT PROGRAM

MISCELLANEOUS PAPER W-92-3

WATER QUALITY '92
PROCEEDINGS OF THE 9TH SEMINAR

16-20 March 1992
San Antonio, Texas

October 1992
Final Report

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Washington, DC 20314-1000

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The U.S. Army Corps of Engineers 9th Seminar on Water Quality was held in San Antonio, Texas, on 16-20 March 1992, to provide an opportunity for presentation of water quality and water control work. Papers presented at the meeting are included in this report.

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CONTENTS

PREFACE

The U.S. Army Corps of Engineers 9th Seminar on Water Quality was held in San Antonio, Texas, on 16-20 March 1992. The seminar was co-sponsored by the Committee on Water Quality (CWQ), the U.S. Army Engineer Waterways Experiment Station (WES), and the Hydrologic Engineering Center (HEC) of the U.S. Army Corps of Engineers Water Resources Support Center.

The organizational activities were carried out under the general supervision of Mr. J. L. Decell, Program Manager, Environmental Resources Research and Assistance Programs (ERRAP), Environmental Laboratory (EL), WES. Mr. Robert C. Gunkel, Assistant Program Manager, ERRAP, was responsible for planning the meeting. Dr. John Harrison was Director, EL, WES. Mr. Pete Juhle was Chairman, CWQ, for the Headquarters, U.S. Army Corps of Engineers.

The seminar agenda was prepared by Mr. R. G. Willey of the HEC. Mr. Robert C. Gunkel and Mr. Thomas R. Patin, EL, WES, were responsible for coordinating the necessary activities leading to publication.

At the time of publication of this report, Director of WES was Dr. Robert W. Whalin. Commander was COL Leonard G. Hassell, EN.

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AGENDA
9th Water Quality Seminar
San Antonio, Texas
16-20 March 1992

MONDAY, 16 MARCH 1992

1:00- 5:00 p.m.	REGISTRATION (Prefunction Area)

TUESDAY, 17 MARCH 1992

7:30- 8:00 a.m.	REGISTRATION (Prefunction Area)
8:00 a.m. -12 noon	PLENARY SESSION (Ballroom A) Chair: Pete Juhle, Headquarters (HQUSACE)
8:00 a.m.	Opening "Kickoff" * Pete Juhle, HQUSACE & Art Denys, Southwestern Division
8:30 a.m.	Committee on Water Quality * Pete Juhle, HQUSACE
9:00 a.m.	DOTS Update * Robert Engler, Waterways Experiment Station (WES)
9:30 a.m.	BREAK
10:00 a.m.	WOTS/WQRP Update * Lewis Decell, WES
10:30 a.m.	Implementation of Environmental Initiatives: WRDA to Funded Wetlands and Habitat Restoration Projects * Carol A. Coch and Marshall G. Nelson, North Atlantic Division
11:15 a.m.	Group Discussion with HQUSACE * Earl Eiker, HQUSACE
12 noon	LUNCH
1:00- 5:00 p.m.	CONCURRENT SESSIONS
	Concurrent Session A: Reservoir & Riverine (Ballroom A) Chair: Warren Mellema, Missouri River Division
1:00 p.m.	Lower Missouri River Water Quality Study * Dorothy Hamlin Tillman, WES

1:30 p.m.	Downstream Water Quality Patterns in Relation to Multilevel Peaking Hydropower Releases from West Point Dam, Georgia-Alabama * William Jabour, WES
2:00 p.m.	J. Strom Thurmond (Clarks Hill) Lake: A Historical Perspective Based on Temperature and Dissolved Oxygen * Joe H. Carroll and John J. Hains, WES
2:30 p.m.	BREAK
3:00 p.m.	Fish Spill and Dissolved Gas Saturation at Columbia River Basin Hydroelectric Dams * A. Rudder Turner, Jr., North Pacific Division
3:30 p.m.	Temporal and Spatial Variability of Riverine Water Quality in a Rural, Agricultural Drainage Basin * Steven L. Ashby, WES
4:00 p.m.	Water Quality Database Management * Henry C. Jackson, Ohio River Division
4:30 p.m.	Panel Discussion (all speakers)

Concurrent Session B: Coastal & Estuarine (Ballroom B)
Chair: Jan Miller, North Central Division

1:00 p.m.	ARCS Program Overview * Paul Horvatin, U.S. Environmental Protection Agency (EPA)
1:20 p.m.	Toxicity Screening/Evaluation * Richard Fox, EPA
1:40 p.m.	Risk Assessment/Modeling * David Cowgill, EPA
2:00 p.m.	Remediation Technologies * Steve Yaksich, Buffalo District (NCB)
2:15 p.m.	Treatment Demonstrations - Buffalo River * Tom Kenna, NCB
2:25 p.m.	Treatment Demonstrations - Saginaw River * Jim Galloway, Detroit District (NCE)
2:35 p.m.	Treatment Demonstrations - Grand Calumet River * Jay Semmler, Chicago District
2:45 p.m.	BREAK
3:15 p.m.	Sediment Quality Criteria: Their Utility as a Tool in Sediment Evaluation * James M. Brannon, Francis J. Reilly, Cynthia B. Price, Judith C. Pennington, and Victor A. McFarland, WES
3:45 p.m.	Considerations in Implementing Cleanup Dredging Under Section 312 of WRDA 90 * Michael R. Palermo, WES, Tom Chase, San Francisco District (SPN), and Joe Wilson, HQUSACE
4:15 p.m.	Overview of Studies on the Adsorption/Desorption of Contaminants from Sediments in Corps of Engineers Reservoir Projects * Douglas Gunnison, Judith C. Pennington, Thomas C. Sturgis, Carlos Ruiz, and James M. Brannon, WES
4:45 p.m.	Panel Discussion (all speakers)
5:00- 6:30 p.m.	POSTER SESSION AND RECEPTION (Ballroom C)

WEDNESDAY, 18 MARCH 1992

8:00 a.m. 12 noon	CONCURRENT SESSIONS
	Concurrent Session A: Reservoir & Riverine (Ballroom A) Chair: Jim Farrell, Lower Mississippi Valley Division
8:00 a.m.	Assessment of Reservoir Water Quality Patterns Using Landsat Images * R. H. Kennedy and J. H. Carroll, WES, B. I. Naugle, Murray State University, and D. Findley, Mobile District
8:30 a.m.	A Water Quality Program for Practical Engineers Under Strong Funding Constraints * M. Markowitz, South Pacific Division
9:00 a.m.	Model Development of Gas Transfer from Bubble Plumes * Steven C. Wilhelms and Richard E. Price, WES
9:30 a.m.	BREAK
10:00 a.m.	Gas Transfer at Low-Head Hydraulic Structures * Steven C. Wilhelms, WES and John S. Gulliver, University of Minnesota
10:30 a.m.	Development of an Expert Advisor for Selective Withdrawal Operations * J. P. Holland, WES
11:00 a.m.	Analysis of Buoyant Flows in Density Stratified Impoundments * Mike Schneider, WES
11:30 a.m.	Panel Discussion (all speakers)
	Concurrent Session B: Coastal & Estuarine (Ballroom B) Chair: Robert Engler, WES
8:00 a.m.	Short- and Long-Term Water Quality Impacts from Riverine Dredging * David L. Wallace, Vicksburg District (LMK)
8:30 a.m.	Analyzing "Less Than" Data: An Example from Sediment Bioaccumulation Testing * Joan U. Clarke, WES
9:00 a.m.	Case Study - A Dispersion Analysis of the Charleston, South Carolina, Ocean Dredged Material Disposal Site * James R. Tallent and Norman W. Scheffner, WES
9:30 a.m.	BREAK
10:00 a.m.	San Francisco Bay Dredged Material Disposal Management * T. H. Wakeman and T. C. Chase, SPN
10:30 a.m.	Dioxin "Toxic Equivalents"--Scientific Regulation or Alchemy?
*	Victor A. McFarland, Joan U. Clarke, Susan A. Jarvis, Charles H. Lutz, Brian Mulhearn, and Francis J. Reilly, Jr., WES
11:00 a.m.	Ocean Disposal of Dredged Material * Thomas D. Wright, WES
11:30 a.m.	Panel Discussion (all speakers)
12 noon	LUNCH

1:00- 5:00 p.m.	CONCURRENT SESSIONS
	Concurrent Session A: Reservoir & Riverine (Ballroom A) Chair: John Andersen, Omaha District
1:00 p.m.	Economic Benefits of Fishery Improvements Associated with Lake Destratification * Richard E. Punnett, Huntington District, and Michael E. Hoeft, West Virginia Division of Natural Resources
1:30 p.m.	Nutritional Implications of Convective Hydraulic Circulation * John W. Barko and William F. James, WES
2:00 p.m.	Sedimentation Dynamics in Eau Galle Reservoir, Wisconsin * William F. James and John W. Barko, WES
2:30 p.m.	BREAK
3:00 p.m.	Little Missouri River Environmental Enhancement Project * David R. Johnson, LMK
3:30 p.m.	Applications of the Cumberland Basin Reservoir System Model for Water Quality Control * Jackson K. Brown, Nashville District
4:00 p.m.	Management Technique for Long-Term Flow Augmentation * Kenneth S. Lee, Baltimore District
4:30 p.m.	Panel Discussion (all speakers)
	Concurrent Session B: Coastal & Estuarine (Ballroom B) Chair: Chuck Wener, New England Division
1:00 p.m.	Initial Development of a Chronic Sublethal Bioassay for the Evaluation of Dredged Material * T. M. Dillon, D. W. Moore, and A. B. Gibson, WES
1:30 p.m.	Combined Hydrodynamic and Water Quality Modeling of Lower Green Bay * Norman W. Scheffner, Barry W. Bunch, David J. Mark, and Keu W. Kim, WES
2:00 p.m.	Overview of Chesapeake Bay Three-Dimensional Water Quality Model * Carl F. Cerco and Thomas M. Cole, WES
2:30 p.m.	BREAK
3:00 p.m.	Dissolved PCB in the Saginaw Confined Disposal Facility During Dredged Material Disposal * Tommy E. Myers, WES, and Pam Bedore, NCE
3:30 p.m.	Bioaccumulation Evaluation of Sediments Using the Guidance Provided in "Evaluation of Dredged Material Proposed for Ocean Disposal" ("The Green Book") * Francis J. Reilly, Jr., Victor A. McFarland, Joan U. Clarke, A. Susan Jarvis, Charles H. Lutz, and J. Brian Mulhearn, WES
4:00 p.m.	Application of Simulated Acid-Rain as an Alternative TCLP Extraction Medium * Frank Snitz, NCE
4:30 p.m.	Panel Discussion (all speakers)

5:00- 5:30 p.m.	GROUP DISCUSSION - JOINT SESSION (Ballroom A) Chair: Pete Juhle, HQUSACE

THURSDAY, 19 MARCH 1992

8:30 a.m. -12 noon	WATER QUALITY RESEARCH PROGRAM FY 93 Civil Works R&D Program Review (Corps of Engineers Representatives Only) (Ballroom A)
1:00- 5:00 p.m.	USE OF WATER QUALITY MODELS WORKSHOP (Renaissance)
1:00- 5:00 p.m.	SEDIMENT-WATER INTERACTIONS WORKSHOP (Ballroom B)

FRIDAY, 20 MARCH 1992

8:00 a.m. - 12 noon	USE OF WATER QUALITY MODELS WORKSHOP (Renaissance)
8:00 a.m. -12 noon	SEDIMENT-WATER INTERACTIONS WORKSHOP (Ballroom B)

POSTER PRESENTATIONS

Acute Toxicity of Illinois River Suspended Sediment to Two Species of Freshwater Animals, *Pimephales promelas* and *Daphnia magna*
* A. B. Gibson, T. M. Dillon, D. W. Moore, and C. A. Beckert, WES

An Evaluation of Sediment-Related Factors Influencing Wintertime Dissolved Oxygen Levels in the Big Eau Pleine Reservoir, Wisconsin
* D. Gunnison, W. F. James, and J. W. Barko, WES

Sediment Oxygen Demand and Its Effects on Water Quality in Corps of Engineers Projects
* Cynthia B. Price, Douglas Gunnison, and Carl Cerco, WES

Application and Use of RECOVERY
* Carlos E. Ruiz, WES

Variations of Oxygenation Efficiency: Concepts and Performance in Richard B. Russell Lake
* John J. Hains and Joe H. Carroll, WES

Zebra Mussel Invasion
* Andrew Miller, WES

History and Rehabilitation of Richard B. Russell Oxygen System
* Diane Hampton, Savannah District

Field Data Input and Sample Tracking
* Bill Easley, Louisville District

Integration of GPS/GIS Technologies
* Charles Hahn, WES

Environmental Resources Research and Assistance Programs
* Bob Gunkel and Andy Anderson, WES

Managing Contaminated Sediments
* Tom Patin, WES

Wetlands Research Program
* Glenn Rhett, WES

NOTE: Posters available for viewing--Tuesday, 17 March (8:00 a.m. - 6:30 p.m.) and Wednesday, 18 March (8:00 a.m. - 5:30 p.m.).

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CONVERSION FACTORS, NON-SI TO SI UNITS OF MEASUREMENT

Non-SI units of measurement used in this report can be converted to SI units as follows:

Multiply	By	To Obtain
acres	4,046.873	square meters
acre-feet	1,233.489	cubic meters
cubic feet	0.02831685	cubic meters
cubic yards	0.7645549	cubic meters
fathoms	1.8288	meters
feet	0.3048	meters
gallons (U.S. liquid)	3.785412	liters
inches	2.54	centimeters
miles (US statute)	1.609347	kilometers
pounds (mass)	0.4535924	kilograms
square miles	2.589998	square kilometers

WATER QUALITY '92
9th Seminar on Water Quality

Introduction

The Corps of Engineers Seminar on Water Quality is held biennially to provide for professional presentation of current research projects and operations activities. Subsequent to these presentations, the Civil Works Research and Development Program Review for the Water Quality Research Program is held. This seminar and program review are attended by representatives of the Headquarters, U.S. Army Corps of Engineers (CE); CE Laboratories and Field Operating Activities; and CE Division and District offices.

The overall objective of this biennial seminar is to provide an opportunity for presentation of water quality assessment, prediction, and control for reservoirs and inland waterways as well as coastal and estuarine water resource projects.

The printed proceedings of the biennial seminar are intended to provide all levels of Corps management with a summary of current water quality research and development as well as operational activities.

The contents of this report include the presentations of Water Quality '92, the Corps of Engineers 9th Seminar on Water Quality, held in San Antonio, Texas, 16-20 March 1992. Abstracts have been substituted for papers not provided.

Predicted Water Quality Impacts from Reducing Navigation Flow on the Missouri River

by
Dorothy H. Tillman[1] and Mark S. Dortch[1]

Background

In recent years the Missouri River basin has experienced a moderate to severe drought that has affected users of the system on the uppe and lower portions of the basin. Because the economies of the upper and lower basin employ different uses of the Missouri River (i.e., recreation in the upper basin has become a large industry, while the lower states are strictly interested in navigation), the operation of the main stem system has become a major concern. Not only have the general public, private industries, and public- and private-owned utilities become concerned about the operations of the Missouri River system, but Federal and state agencies are also concerned with river operations.

All these concerns have prompted the Missouri River Division (MRD) to reevaluate the operations of the Missouri River system and initiate a revision of the Master Water Control Manual. The update to the Master Manual was to be completed in two phases. Phase 1, which has been completed, reviewed present operations criteria from the current Master Water Control Plan over the period of record (1898 to the present) and compared it to alternative water control plans. Phase 2 (whose plan was formulated based on Phase 1) concentrated on environmental studies required by the National Environmental Policy Act and all other environmental laws. From Phase 2 results, a water control plan will be selected to be included in the revised Missouri River Master Water Control Manual.

Study Objective

The Environmental Laboratory of the US Army Engineer Waterways Experiment Station was requested to assist the MRD in the numerical modeling of a number of water quality constituents in the lower Missouri River for Phase 2 of the Master Water Control Manual Review and Update. Model results were used to determine the effects on key water quality constituents (i.e., temperature and dissolved oxygen) from reducing the historical seasonal navigation releases for a range of release temperatures at Gavins Point Dam.

Site Description

The study reach of interest for this study extends approximately 800 miles[2] from Gavins Point Dam to the Mississippi River. From the mouth of the Missouri River to Sioux City, IA (732 miles), regulation is managed by structural measures (i.e., dikes, revetments, and sills) and partial reservoir control of streamflow to provide a navigable channel (Slizeski, Andersen, and Dorough 1982). Commercial navigation above Sioux City is not attempted because of the shallower depths and the six main stem dams.

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[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

[2] A table of factors for converting non-SI units of measurement to SI units is presented on page xxii.

Releases during non-navigation season range from 15,000 to 20,000 cfs. During navigation season (March through October), the releases are maintained in the range of 30,000 to 35,000 cfs. Discharges from the tributaries during navigation season can add an additional 15,000 to 20,000 cfs between Sioux City and the mouth of the Missouri River.

Average velocities in the Missouri River range from 3 to 6 fps, and during navigation season, midchannel velocities are in the range of 4 to almost 7 fps (MRD 1978). The slope of the bed is approximately 1 ft per mile. In the 1960s and 1970s, a lowering of the riverbed occurred, but this has stabilized except for the reach between Sioux City and Blair, NE, which is still experiencing bed lowering (MRD 1978).

General Modeling Approach

Changes in water quality were assessed by numerically modeling the Missouri River from Gavins Point Dam to the junction of the Mississippi River (approximately 800 miles). The US Environmental Protection Agency's (USEPA) one-dimensional (longitudinal) riverine model, QUAL2E, was the model used for the study.

A steady-flow, steady-state water quality modeling approach was selected for this study for several reasons. During a drought period, a reasonable assumption was that tributary inflows would be essentially constant from lack of rainfall. Likewise, release flows from Gavins Point Dam during navigation season are relatively constant as well. Thus, the assumption of steady flow is a reasonable assumption. Steady-state loadings are usually associated with steady flow. In addition, issues addressed in this study are similar to those in wasteload allocation studies. For instance, in a wasteload allocation study, pollutant loadings within a stream are modeled to determine the impact on instream water quality. In this study, release flows from Gavins Point Dam would be altered (e.g., reduced) instead, with wasteloads unmodified, to determine impacts on water quality. For riverine water quality model studies of this type, the assumption of steady-state conditions is usually made and is an acceptable approach. Finally, steady-state models require far less data and effort to calibrate and verify than are required for dynamic (i.e., time-varying) models. For example, application of a dynamic model to the lower Missouri River would require time-varying water quality boundary conditions and instream observations (for calibration/verification) for at least a month-long period. Thus, approximately daily (or every few days), monitored data would be required for all major tributaries and instream Missouri River stations (about every 20 miles). The cost to accomplish a data collection effort of this magnitude would have been excessive and unjustified. When conditions are near steady state, snapshot sampling (i.e., collection of data at all stations in a relatively short period of time, such as a day or two) can be used to support a steady-state model with much less cost.

Calibration and verification of QUAL2E were completed using data collected by the Corps of Engineers Omaha and Kansas City Districts in August and September 1990. Once calibration and verification of the model were satisfactorily complete for the Missouri River, scenario runs requested by the MRD were simulated. These scenario runs evaluated varying release flow and temperatures from Gavins Point Dam. Sensitivity analyses were also performed to evaluate the effects of varying flows and water quality concentrations from the tributaries, meteorological conditions, and power plant loads.

Model description

QUAL2E is a one-dimensional riverine water quality model with the capability of simulating up to 15 water quality constituents of any branched stream. Constituents that can be modeled in any combination by the user are listed below (Brown and Barnwell 1987).

a. Dissolved oxygen (DO).

b. Carbonaceous biochemical oxygen demand (CBOD).

c. Temperature.

d. Algae as chlorophyll *a*.

e. Organic Nitrogen as N.

f. Ammonium as N.

g. Nitrite as N.

h. Nitrate as N.

i. Organic phosphorus as P.

j. Dissolved phosphorus as P.

k. Coliforms.

l. Arbitrary nonconservative constituents.

m. Three conservative constituents.

The above constituents can be simulated in a steady-state mode (the time derivative of concentration is omitted from the mass balance equation) or dynamic mode (meteorological data can change with time). The model is based on the time-dependent water quality constituent transport equation, allowing for description of advection, dispersion, and sources/sinks. This equation is referred to as the energy equation for temperature or the differential mass balance equation for other constituents.

Hydraulic conditions (flow rate, velocity, and depth) used within the energy and mass balance equations are determined from steady, nonuniform flow conditions by satisfying continuity and using stage-discharge relationships or solving Manning's equation with channel geometry information. Steady flow implies that the flow, velocity, width, and depth at a given point in the stream network are constant with time. Nonuniform flow allows velocity, flow, width, and depth to change in the longitudinal direction from reach to reach.

QUAL2E approximates the river system by subdividing the stream system into reaches (the basic division of the model). Reaches represent portions of the river having similar channel geometry, hydraulic characteristics, and chemical/biological coefficients. Reaches are further divided into equally spaced units called computational elements. Each computational

element has inputs, outputs, and reaction terms. The energy and differential mass balance equations are solved simultaneously (implicitly) for computational elements.

Model modifications

Modifications to the QUAL2E code were necessary to accomplish study needs and improve model performance. Modifications to the code were (a) allowing the rating curves used to calculate depths and velocities to be read per element rather than per reach, (b) increasing number of point sources allowed, (c) modifying the read statement for hydraulic data to also include input values for delta temperatures and discharges coming from power plants, (d) adding heat source from power plants to the temperature equation as a separate term, (e) adding contribution of algae that enters into organic carbon as CBOD, and (f) adding the temperature correction term (rate multiplier) for algae used in the CE-QUAL-R1 model (Environmental Laboratory 1982).

Observed data

Numerical models require many types of observed data to adequately model a water body system. The types of data required to calibrate/verify QUAL2E for the lower Missouri River study were:

a. Hydraulic data or channel geometry/flow conditions.

b. Headwater boundary conditions.

c. Point source and tributary loads.

d. Meteorological data.

e. Rate coefficients.

f. Calibration/verification comparison data.

To provide enough data for calibration/verification, the Omaha and Kansas City Districts conducted two snapshot samplings (taking measurements at all stations within a day) of water quality concentrations on the main stem Missouri River and 19 major tributaries. Tributary data were used for headwater and tributary boundary conditions during calibration and verification. Main stem data were used for model comparisons. The Omaha and Kansas City Districts also provided hydraulic data from HEC-2 (Hydrologic Engineering Center 1982) simulations which were used in developing rating curves for QUAL2E. Point source data (e.g., wastewater treatment plants) were obtained from the Region VII USEPA office in Kansas City, KS. Meteorological data for four first-order meteorological stations (Sioux City, IA; Omaha, NE; Topeka, KS; and Columbia, MO) were obtained from the US Air Force Environmental Technical Applications Center in Asheville, NC.

QUAL2E Calibration/Verification

Model calibration is an iterative process that requires comparison of model output to observed data for refining and adjusting model parameters until optimal model predictions are obtained. Water quality model calibration is actually a two-step process. First, calibrated

hydraulic conditions must be in agreement with observed conditions. After the model is hydraulically calibrated, water quality calibration is performed until water quality predictions are in agreement with the observed water quality values. Once the calibration process is completed, a second data set, preferably with different flows and loadings, is used to verify that the model produces acceptable results. All model parameters (i.e., coefficients) remain the same. A calibrated and verified model can then be used to determine the effects of operational changes on downstream water quality.

Calibration

Hydraulic calibration of QUAL2E was not necessary since hydraulic data were furnished by the Omaha and Kansas City Districts and provided to the model. The hydraulic data came from HEC-2 simulations, where HEC-2 had been calibrated against observations. The HEC-2 results were used to develop (through regression) rating curves required by QUAL2E to compute, from discharge, depths and velocities needed at computational elements. Computed depths were compared to the HEC-2 results to see how accurately the rating curves estimated depths. On the average, computed depths were about 86 percent of the HEC-2 simulated depths for four locations examined. This accuracy was considered acceptable.

Calibration was performed on all water quality constituents of concern and compared with observed Missouri River data. Water quality data available for calibration included temperature, DO, CBOD, chlorophyll *a*, organic nitrogen, ammonia nitrogen, nitrate nitrogen, and dissolved inorganic phosphorus. Data collected during the snapshot sampling on August 28, 1990, by the Omaha and Kansas City Districts or reported by power plants were used in the calibration runs as tributary and headwater boundary conditions and power plant loads, respectively. In addition, measured data collected August 28, 1990, on the main stem Missouri River were used as comparison data to evaluate model predictability.

Model performance was also evaluated using the mean absolute error (MAE) and the root mean square error (RMSE). The MAE represents the average error ($\pm$) in model predictions as compared to observed data, and the RMSE is a measure of variability between predicted and observed values.

Model calibration consisted of adjusting coefficients and comparing the predicted and observed concentrations. This was an iterative process that was repeated until reasonably close comparisons between observed and predicted concentrations were made. Reaction rates for processes (e.g., algal growth and respiration, organic nitrogen hydrolysis, nitrification, CBOD oxidation, etc.) were initially set based on recommendations by Brown and Barnwell (1987), with the exception of CBOD. The initial oxidation rate for CBOD was calculated using linear regression (PROC REG, SAS Institute 1988) from 20-day CBOD with daily analysis at four river mile (RM) locations (RM 735, 534, 329, and 50). All estimated K's were similar in value, so an average value of 0.12 having a R-square of 0.93 for the regression was set for all reaches.

Initially, algae was not included as a modeled constituent since many people were of the opinion that very little algae existed in the Missouri River because of faster flow velocities and high turbidity. However, this assumption had to be reconsidered after examining initial results for CBOD and organic nitrogen. During initial calibration (Figure 1), it was noted that the dynamics of CBOD (as well as organic nitrogen) were not being predicted correctly. Trends in observed data showed concentrations increasing downstream while predicted data

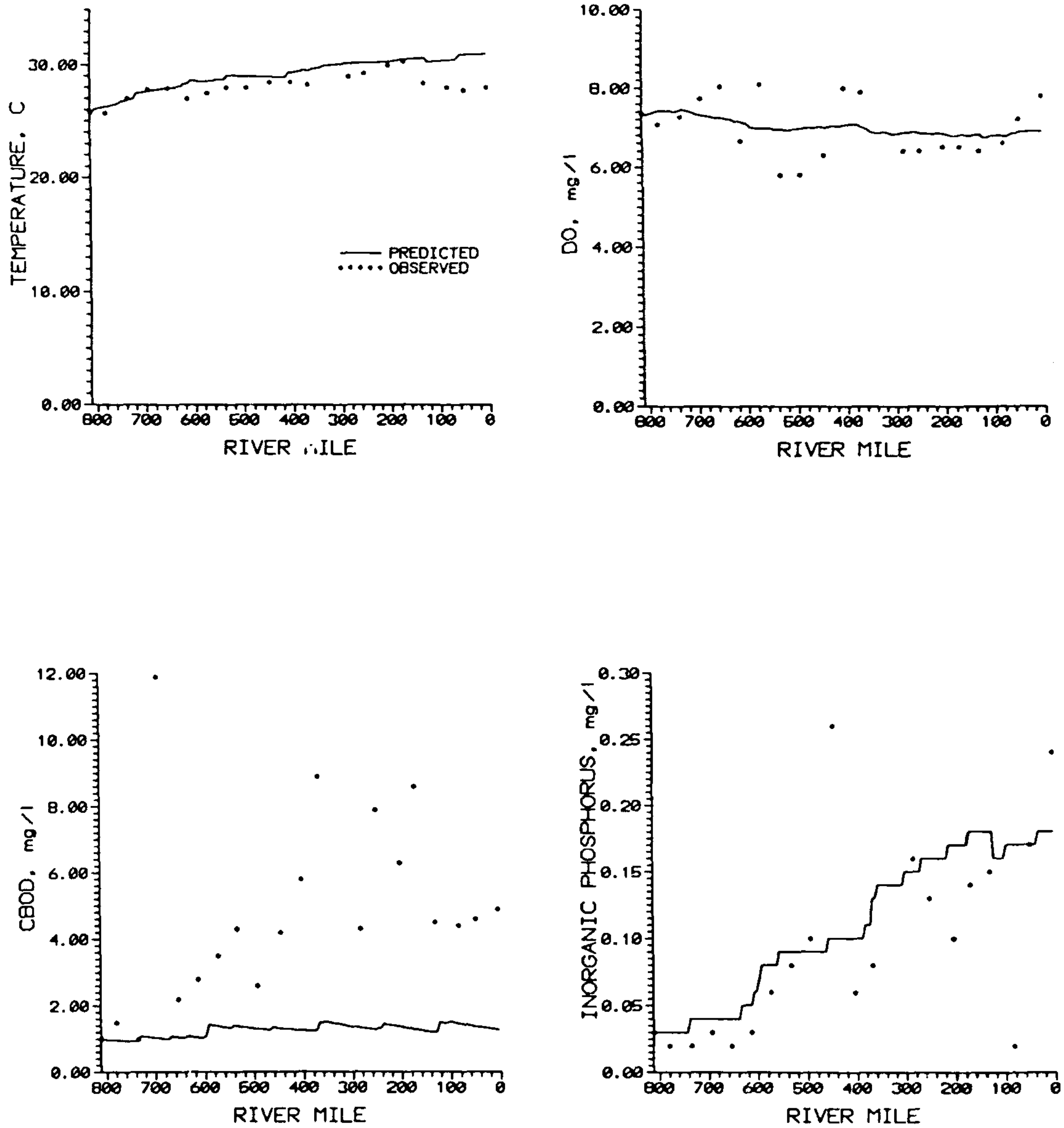


Figure 1. Initial calibration results (without modeling algae) (Continued)

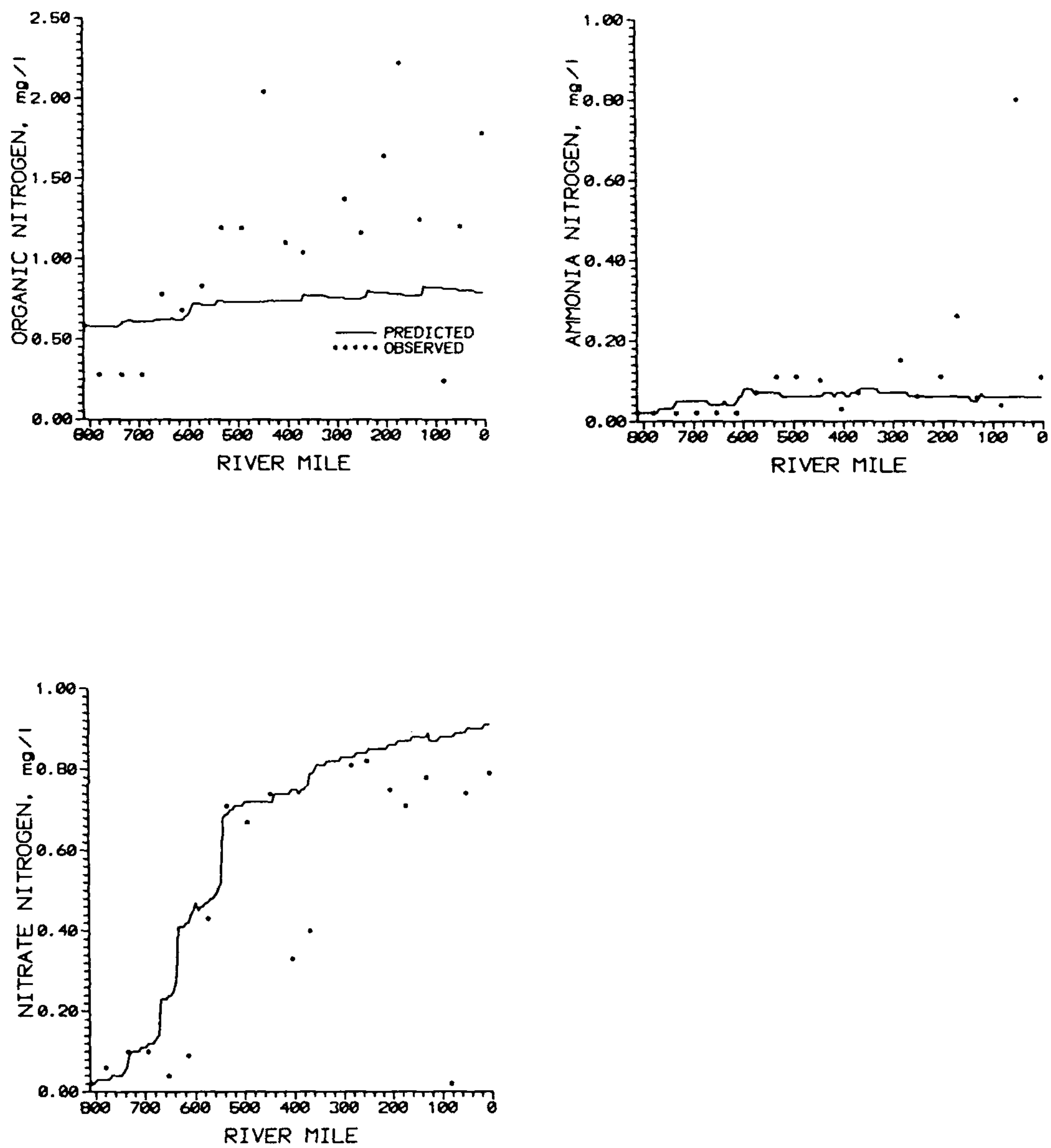


Figure 1. (Concluded)

showed very little increase downstream. Since increases in CBOD loads did not improve predictions, another possible explanation was that algae contributed to CBOD concentrations through algal respiration, mortality, and excretion. After the inclusion of algae as a modeled constituent, final calibration results shown in Figure 2 indicate that the trend of the predicted data was now following the trend of the observed. Figure 2 also presents the results of the other constituents modeled and shows improved predictions over initial calibration results for most of the water quality constituents modeled.

Verification

Model performance was verified using data collected by the Omaha and Kansas City Districts on September 12, 1990, from the second snapshot sampling. Point source boundary conditions were set to data reported in the Permit Compliance System on this date by industries discharging into the Missouri River, and power plant boundary conditions were also set to reported values on this date. Reaction rates for the verification simulation were set to the final values of the reaction rates from the calibration simulation. Figure 3 shows the final verification results for all water quality constituents modeled. Final verification results compared favorably with observed data, as indicated by the MAE and the RMSE (i.e., temperature and DO were 0.64, 0.74 and 0.44, 0.52, respectively).

As shown in Figure 3, most of the observed DO data measured on September 12, 1990, were found to be supersaturated for the water temperatures measured on this date. Why these values were supersaturated was never determined. Because of this, predicted DO concentrations were always approximately 0.4 mg/L lower than observed concentrations, even though predicted water temperatures were close to observed.

Scenario Results

Discussion

Scenario simulations were conducted following model calibration and verification. Scenario runs were made to examine the effects on water quality of operational changes (e.g., reduction in release flow in combination with a band of release temperatures) at Gavins Point Dam and changes to boundary conditions (i.e., meteorological, tributary, point source, and power plant loadings). Eleven sets of scenario runs were made, totaling 123 runs. This paper will present results from five runs from Scenario 1 showing the effects on water quality from reduction of release flows at Gavins Point Dam.

The headwater boundary conditions for release temperature were set to 23.9 °C, and reduced flows were set to 255, 340, 510, and 708 cu m/sec with a base condition release temperature of 25 °C and flow of 779.3 cu m/sec. Base condition is considered a "baseline" for comparison of reduced release flow results. Extreme conditions for tributary, point source, meteorological, and power plant boundary conditions were selected for these scenario runs to induce stress in the Missouri River system. Tributary boundary conditions of flow and water quality concentrations were set to a minimum 7-day running average of flow with a 10-year recurrence period (7Q10) and maximum historical water quality concentrations measured on all tributaries modeled, respectively. Point source (i.e., wastewater treatment plant) boundary conditions were set to maximum allowable permit limits for discharge and water quality concentrations. If a permit did not specify concentration limits for water quality constituents modeled in this study (for example, ammonia), concentrations were set to

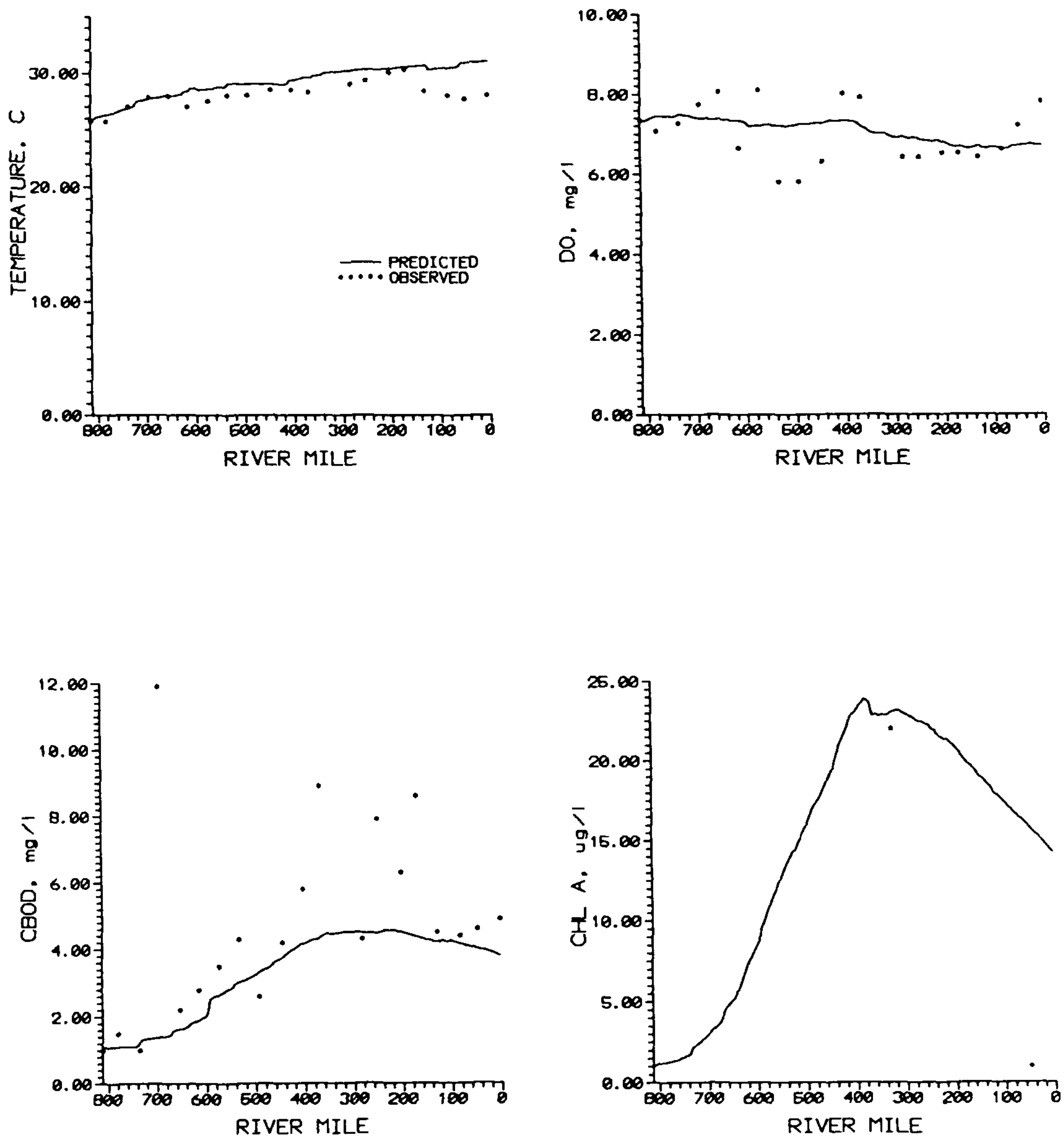
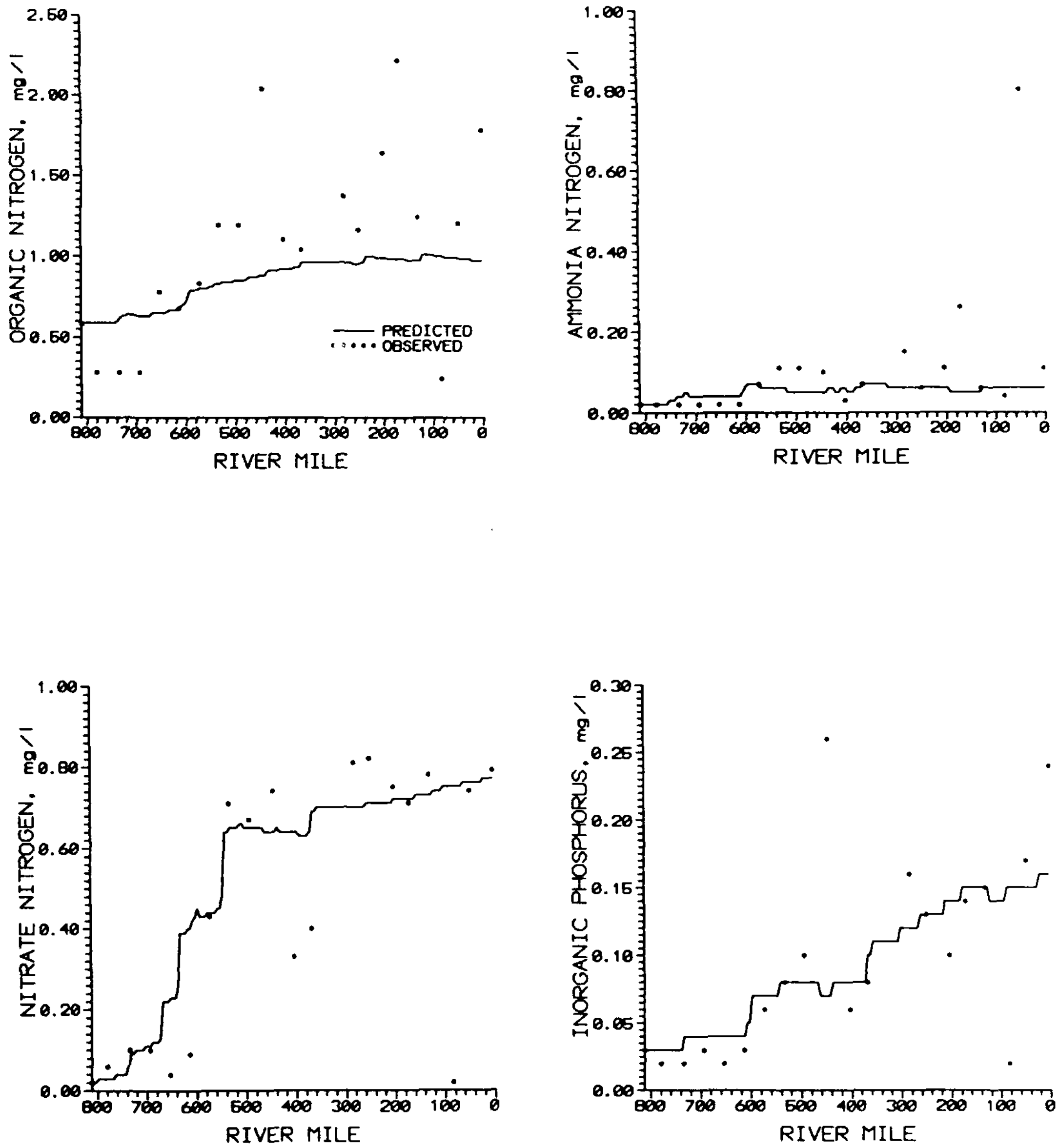


Figure 2. Final calibration results (with algae as a modeled constituent) (Continued)



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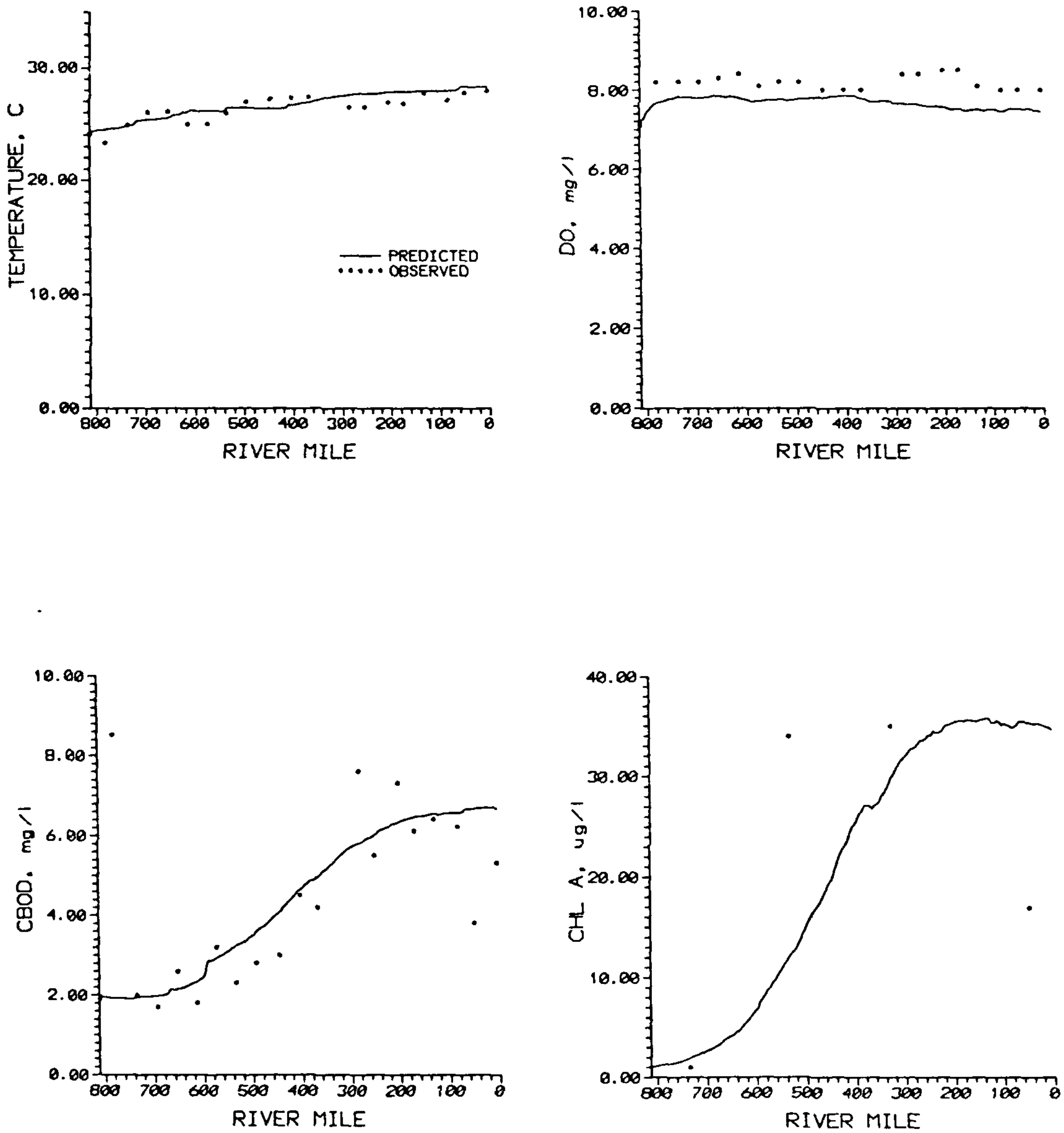

Figure 3. Verification results (Continued)

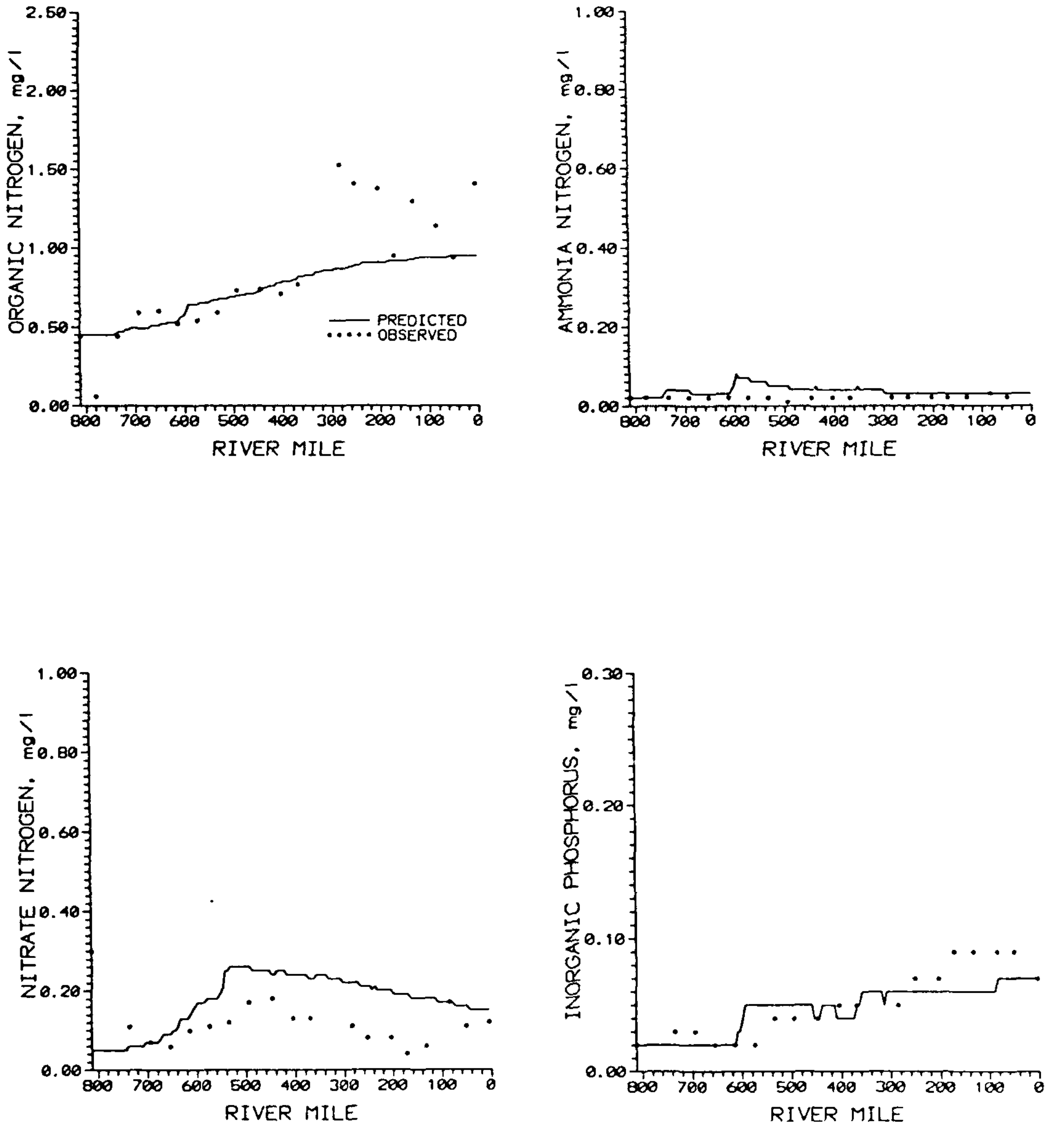

Figure 3. (Concluded)

recommended values from Metcalf and Eddy, Inc. (1979). Meteorological conditions were set to the 14T10, which is defined as the maximum 14-day running average of air temperature that for the summer is expected to occur or be exceeded every 10 years. The 14-day running averages were used because the travel time through the Missouri River system is approximately 2 weeks during navigation season. Power plant boundary conditions for discharge and delta temperature (i.e., differences in effluent and influent power plant cooling water temperature) were set to values reported on August 28, 1990.

Scenario 1

Results (shown in Figure 4) demonstrated the effects of reducing release flow from Gavins Point Dam by comparing the results for release flows of 255, 340, 510.1, and 708 cu m/sec to the base condition results represented by the solid line on all plots (labeled BASE SCEN). Figure 4 shows that by reducing flow, water temperatures were affected, especially at the lower release flows. The greatest temperature difference occurred between the base condition run and the lowest reduced flow (255 cu m/sec), and the maximum difference was approximately 1.3 °C. Temperature results for the other reduced release flows that were analyzed approached base condition temperature results, as the reduced flow value increased to the base condition release flow value. Reduction in release flow also had an effect on the algae concentrations (CHL A in Figure 4) in the Missouri River. Algae concentrations more than doubled in the lower reaches of the Missouri River as the release flow was reduced to 255 cu m/sec (around RM 100). This was attributed, in part, to less dilution of nutrients from tributaries and point sources. At the lower flows, velocity was reduced, increasing residence time in the system, which permitted more algal growth. As algae concentrations increased for all reduced flow runs, concentrations of the other water quality constituents that relate to algae (e.g., CBOD and organic nitrogen) also increased (Figure 4). The greatest concentration increases for these constituents occurred at the lowest reduced release flow when compared to the base condition results. All constituent concentrations doubled or more than doubled at the lowest flow (255 cu m/sec). Some of the increase in these constituent concentrations was attributed to increased algal concentrations, but some can also be attributed to less dilution of tributary and point source loads caused by the reduced release flows.

Even though concentrations of constituents exerting a demand on DO increased as release flows were reduced, DO concentrations in the Missouri River were affected only slightly ($\pm$0.4 mg/L). Increased water temperatures caused DO concentrations to decrease, because increasing water temperatures decrease solubility of DO. DO concentrations probably increased around RM 500 because of increased algal photosynthesis. Water temperatures in this reach did not have much impact on DO concentrations since differences in water temperatures between reduced flow runs were minimal.

Conclusions

From the scenario presented, the following conclusions were derived from an examination of the results:

a. Reduction in release flows out of Gavins Point Dam, from a base condition of 779.3 cu m/sec to 255 cu m/sec, and extreme boundary conditions set for other forcing functions produced increases in water temperatures as much as 1.3 °C. Although water temperatures were increased, their values were well below the 32.2 °C temperature limit for Missouri River water temperatures. Temperature changes in the

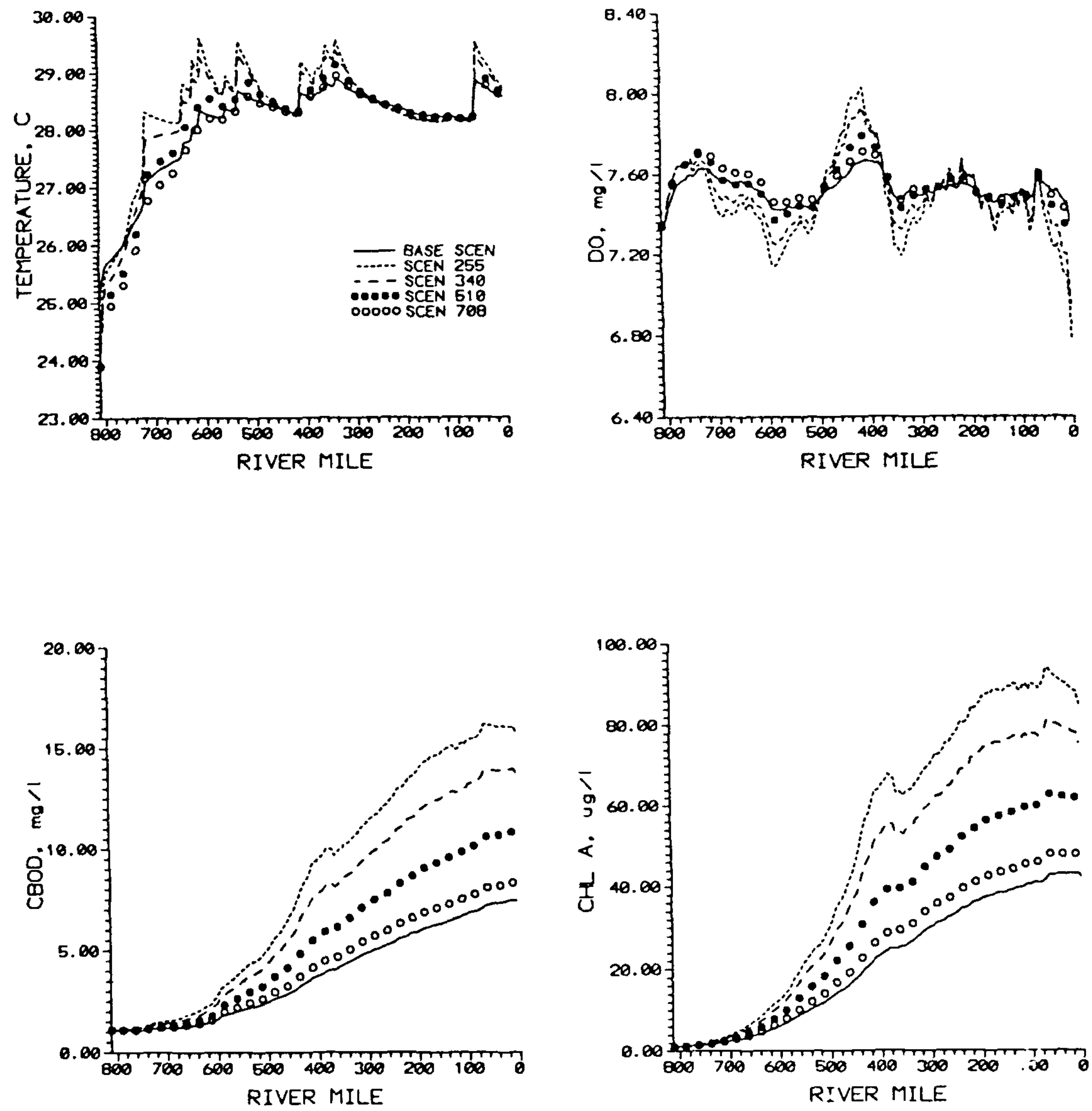

Figure 4. Scenario 1 results (Continued)

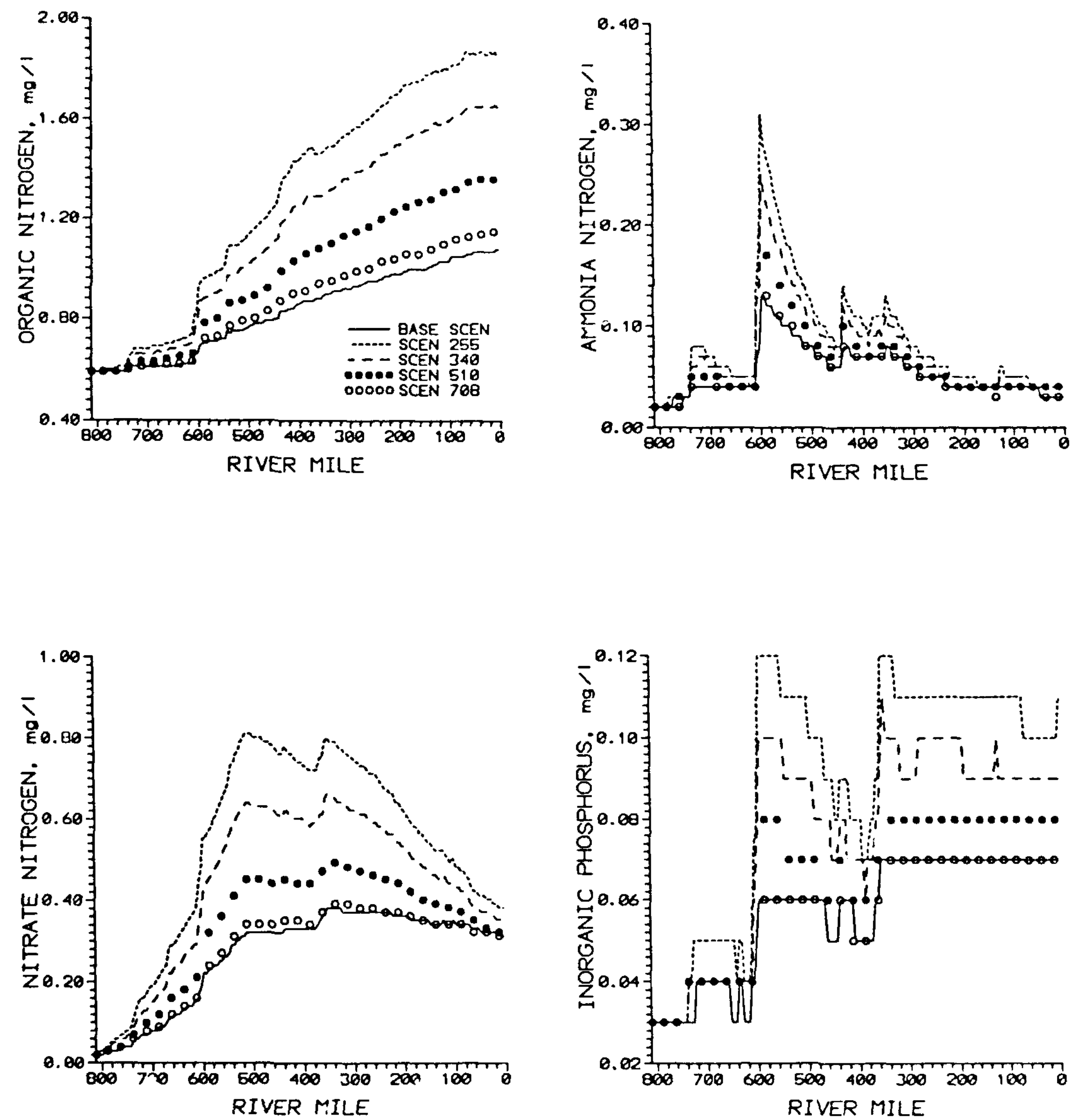

Figure 4. (Concluded)

mixing zone downstream of power plant discharges could not be addressed in this study because QUAL2E assumes inflows to be completely mixed within the channel cross section upon entering.

b. Reduction in release flows out of Gavins Point Dam, from a base condition of 779.3 cu m/sec to 255 cu m/sec, and extreme boundary conditions set for other forcing functions increased all water quality constituent concentrations in the Missouri River except DO concentrations. As release flows were reduced, concentrations of algae, CBOD, organic nitrogen, ammonia nitrogen, nitrate nitrogen, and dissolved inorganic phosphorus were increased twofold or slightly more for some constituents (Figure 4). Most constituent concentrations increased as a result of less dilution of tributary and point source loads, but to some extent, concentrations increased as a result of interactions with algae. Even though concentrations more than doubled in value, none of the State standard limits for these constituents were violated. Reductions in DO concentrations were considered minimal, and were well above State standards for DO.

Overall, results from all scenario runs indicate that water temperature and water quality concentrations were impacted by reducing flows in the Missouri River, yet temperatures and water quality concentrations for all constituents were well within standards for State limits.

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Downstream Water Quality Responses to Hydropower Releases from West Point Dam, Georgia-Alabama

by
William E. Jabour,[1] Steven L. Ashby,[1] and Robert H. Kennedy[1]

Introduction

Reservoir tailwaters, a valuable resource providing water supply, recreational opportunities, and aquatic habitat, are often impacted by hydropower operations. Impacts include changes in flow and physicochemical characteristics. Discharge of hypolimnetic water impacts temperature, dissolved oxygen (DO), turbidity, and chemical processes in the tailwater region. The fate of reduced materials in the discharge is of particular interest since associated reactions influence concentrations of DO, turbidity levels, and other chemical processes. Reaction mechanisms and rates have been described in laboratory studies (e.g., Stumm and Morgan 1981) and lake studies (e.g., Mortimer 1941, 1942; Delfino and Lee 1971); however, studies conducted in reservoir tailwaters have been limited.

A common theme of recent tailwater studies is disparities between field and laboratory observations of reaction rates of reduced materials (Nix 1986, Gordon 1989, Nix et al. 1991). Interactions of substrates in the tailwater region and site-specific characteristics have been suggested as sources for the observed differences. Consequently, delineation of hydropower impacts on these processes is fundamental to provide better management of these resources. The objective of this study was to describe the distribution and fate of selected physicochemical variables in a hydropower discharge for the assessment of operational impacts on tailwater quality.

Site Description

West Point Dam and Lake, located on the Georgia-Alabama border, is a multipurpose project that provides hydroelectric power, flood control, water supply, fish and wildlife development, and flow regulation for downstream navigation. The warm, monomictic reservoir extends approximately 56 km along the Chattahoochee River and has a surface area at full pool of 10,467 ha and a drainage area of 8,754 sq km. Total volume of the reservoir is 45.7 million cu m; estimated retention time is 55 days. West Point Lake typically exhibits thermal stratification with low DO concentrations and high iron and manganese concentrations in the hypolimnion from June through September.

Hydropower capabilities include two main units, each with a capacity of 35 MW, which are operated to meet peak power requirements (peaking generation), and a smaller house unit (low-flow generation), with a capacity of 3.4 MW, which is operated continuously to meet downstream minimum-flow criteria. Discharge rates for the house unit and both main units average approximately 14 and 450 cu m/sec, respectively.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

The tailwater region extends 15.2 km from West Point Dam downstream to a low-head dam located at Langdale, GA. The tailwater provides water to downstream industry and is a source of recreation. During low-flow generation, the tailwater region is a series of pools and riffles that are inundated during peaking generation. In addition to reservoir releases, two secondary tributaries (Oseligee and Long Cane Creeks) contribute inflow to the tailwater.

Methods and Materials

In situ measurements and water samples were collected from a vertical profile to describe water quality conditions in the forebay. Sampling included depths coinciding with penstock openings for the house unit (surface elevation of 194 m National Geodetic Vertical Datum (NGVD) to 184.4 m NGVD) and main generators (182.6 m NGVD to 169.2 m NGVD).

Two methods of sampling were employed to describe water quality conditions at selected sites in the tailwater during normal operations and changes in water quality of releases during peaking generation. The first sampling method involved routine monitoring at fixed stations throughout two periods of peaking and low-flow generation. For this sampling method, five sites were selected along the tailwater from the downstream buoy line (approximately 0.1 km from the dam) to 15.2 km downstream to the low-head dam at Langdale (Stations 10-50, Figure 1). Each station was sampled six times during the 2-day study. Additional sampling was conducted at Stations 10 and 40 to determine the adequacy of the six routine samples. Staff gages were deployed at each station, and water levels were recorded as samples were collected.

The second sampling method involved periodic sampling of a parcel of water as it traveled through the study reach (Stations A-J, Figure 1). Sampling occurred late in the peaking generation cycle and, thus, during near-steady state condition. The moving parcel of water was identified using surface floats. Samples were collected from a boat at approximately equal intervals of distance and referenced to landmarks.

Samples for chemical analyses were placed in acid-washed bottles and preserved with acid or held at 4 °C until analyzed. Turbidity and metals analyses were conducted using standard methods (American Public Health Administration 1985). Filtration employed 0.45-μm filters and was conducted in the field. Temperature, DO, pH, and conductivity were measured in situ with a Hydrolab Surveyor II (Hydrolab Corporation, Austin, TX).

Results and Discussion

Temperature, DO, and metals concentrations varied with depth in the forebay (Figure 2). Water temperatures ranged from 22 to 29 °C, and a surface mixed layer extended to a depth of 5 m. Dissolved oxygen concentrations decreased rapidly below the mixed layer; oxygen concentrations at depths below 10 m were less than 1 mg/L. Iron and manganese concentrations, which were near detection limit (0.05 mg/L) in the mixed surface layer, ranged from 0.3 to 5.0 mg/L and from 0.2 to 2.0 mg/L, respectively, at depths below 10 m. As a result of these gradients, water quality of the releases would be expected to differ during low-flow and peaking generation. Discharge during low-flow generation would be warm (26 to 28 °C), well oxygenated, and would have iron and manganese concentrations near detection limit. Discharge during peaking generation would have lower temperatures (22 to 25 °C) and DO concentrations less than 2.0 mg/L. Iron and manganese concentrations would be above detection limit but less than 5.0 and 2.0 mg/L, respectively.

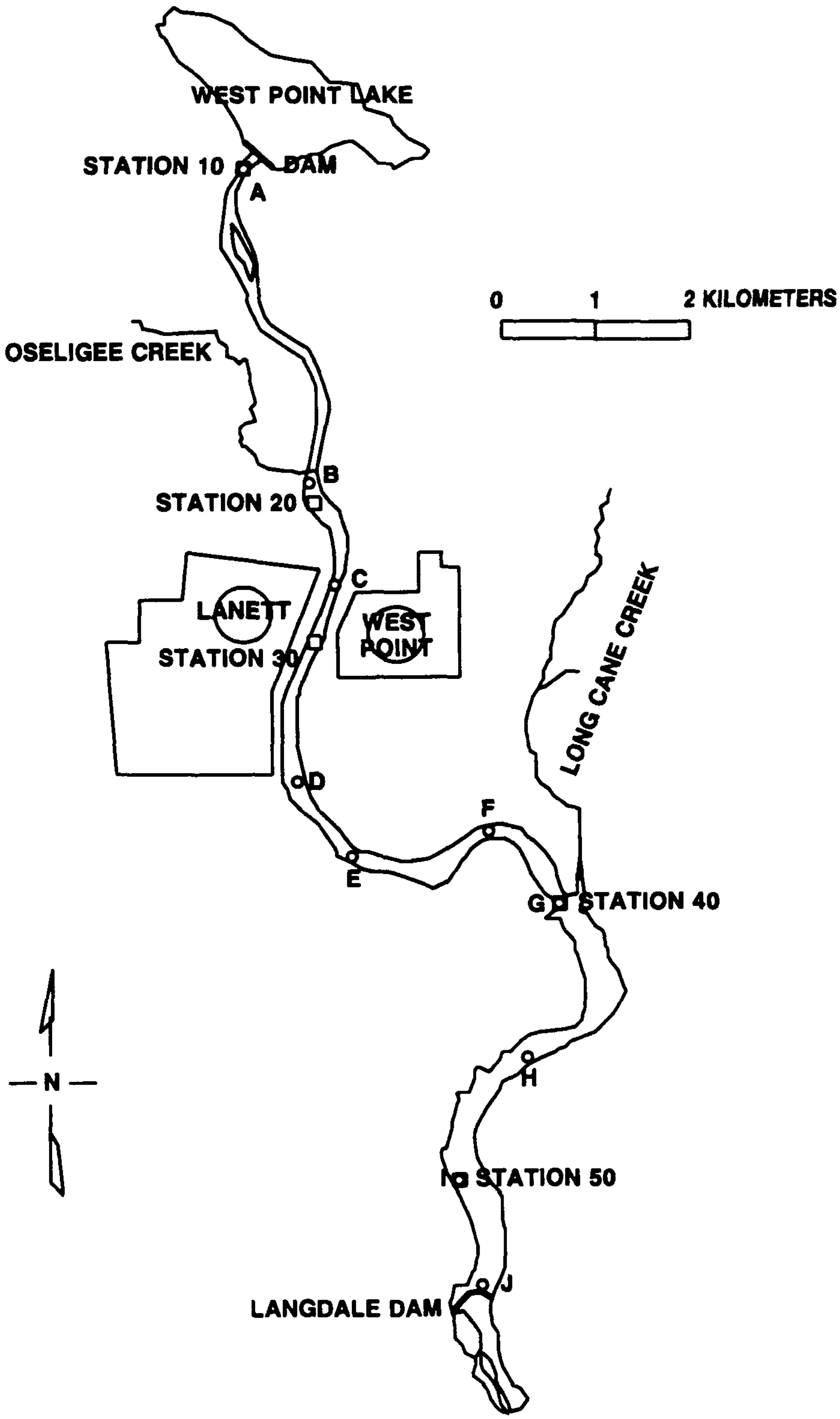

Figure 1. Site map and station locations

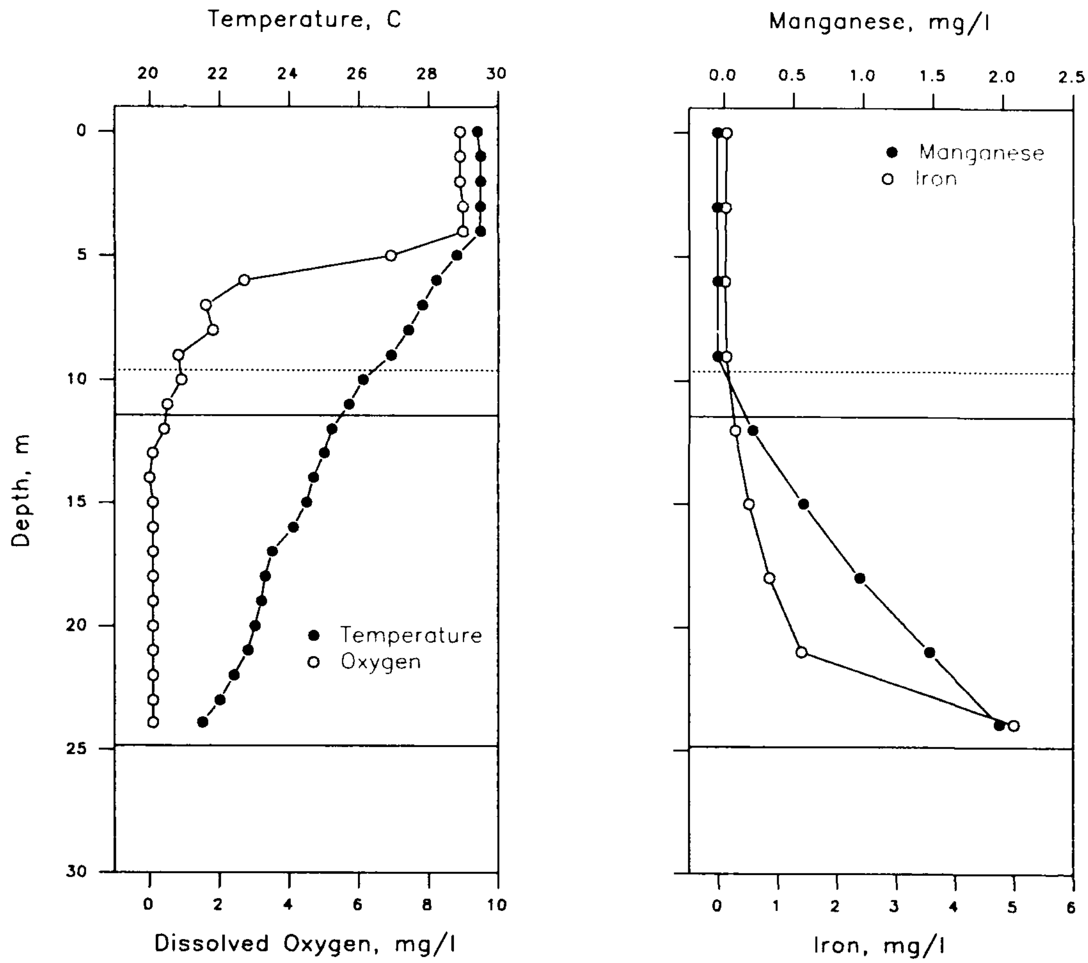

Figure 2. Vertical distribution of temperature, dissolved oxygen, iron, and manganese in the forebay of West Point Lake. Solid lines denote withdrawal zone of main hydropower units; dashed line denotes lower limit of withdrawal for the house unit

Discharge ranged from 14 cu m/sec during low-flow generation to 445 cu m/sec during peaking generation, resulting in changes in stage height of 2 to 2.5 m (Figure 3). Marked increases in discharge velocities and flooding of secondary tributary channels were also observed during peaking generation. Following cessation of peaking generation, stage height rapidly decreased, and pooled water in the secondary tributaries was returned to the main channel.

Influences of hydropower releases were more apparent in the distribution for DO concentrations than for temperature in the tailwater (Figure 3). While temperatures were relatively constant at 27 °C, DO concentrations were approximately 2 mg/L lower during peaking generation than during low-flow generation. Recovery to prepeaking-generation concentrations was observed at Stations 10 and 20 between peaking generation cycles. Maximum DO concentrations at Station 40 occurred coincident with increased flows associated with peaking generation, and increased DO concentrations during low-flow generation were less apparent than observations at upstream stations.

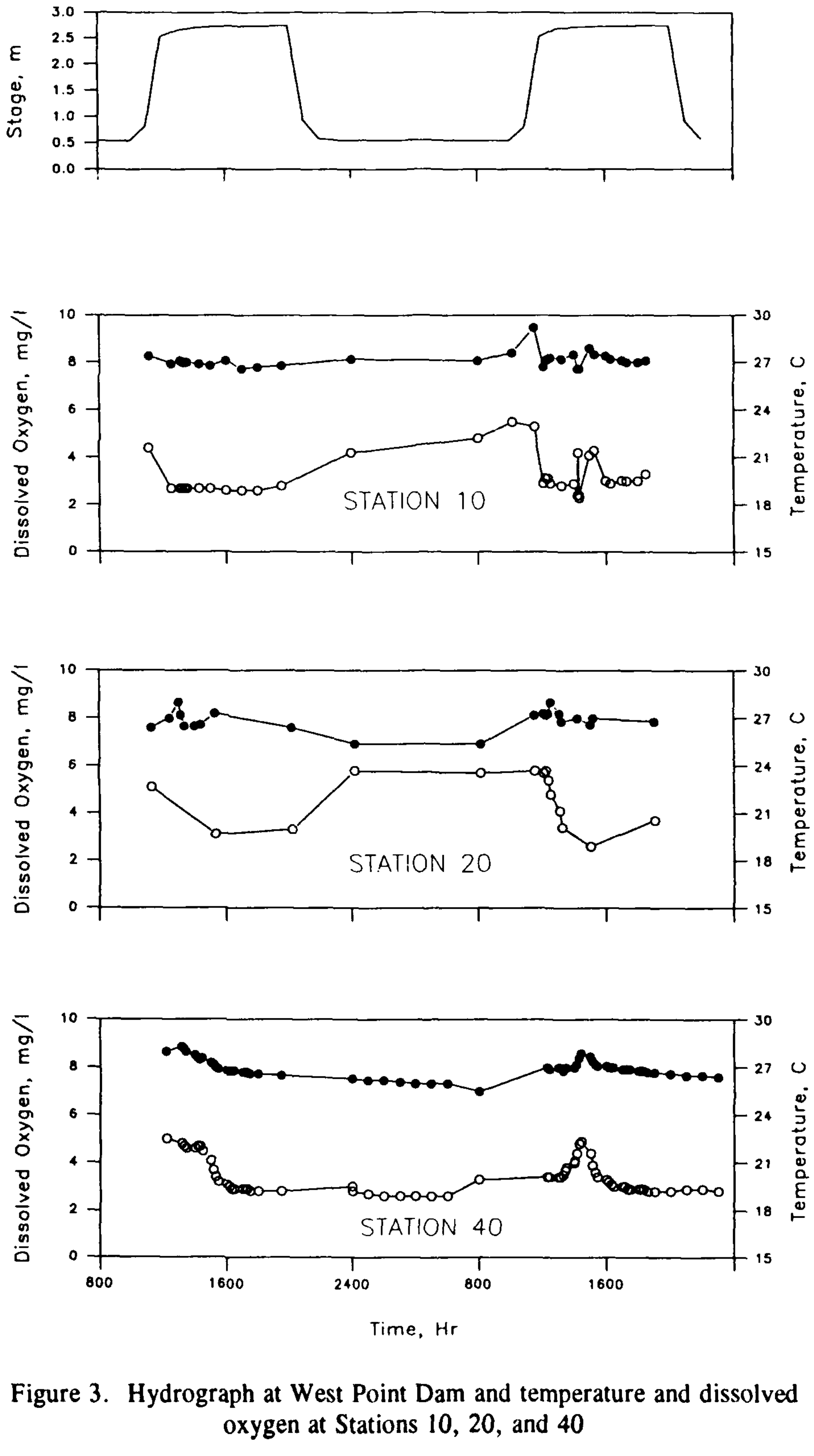

Figure 3. Hydrograph at West Point Dam and temperature and dissolved oxygen at Stations 10, 20, and 40

Observed changes in DO concentrations in the upstream region of the tailwater may be attributed to low DO concentrations in the discharge from West Point Dam during peaking generation, higher DO concentrations in the discharge of surface water during low-flow generation, and aeration (due to riffles in the channel) during low flow. Possible mechanisms leading to observed patterns of DO concentrations at Station 40 were less clear. Decreased DO concentrations suggest oxygen consumption by biological and/or chemical processes. The source of this oxygen demand was not readily apparent. The peak in concentrations may be a function of increased aeration associated with the peaking-generation hydrograph or transport of water with higher DO concentrations from upstream sites.

Patterns of temporal and spatial distribution of manganese are shown in Figure 4. Concentrations at Stations 10 and 20 were relatively constant, and manganese was present primarily in the dissolved form. Dissimilar patterns were observed at Station 40. Total manganese concentrations increased during each peaking-generation cycle, and particulate manganese was present. The increase was short lived, and concentrations diminished rapidly with time. Higher total manganese concentrations suggest sources in addition to the discharge from West Point Dam. The rising limb of the peaking-generation hydrograph results in increased turbulence, sediment resuspension and scour, and increased transport of this material. Settled particulate manganese may be a component of this material and would account for observed increases in total and particulate manganese concentrations. Increased particulate manganese, as a result of oxidation of reduced manganese, is another possible source of particulate manganese and a possible component of oxygen demand mentioned above.

Changes in iron concentration were similar to those observed for manganese. However, particulate iron comprised the majority of the total iron within the tailwater region (Figure 5). Concentrations of total and dissolved iron were relatively constant at Station 10, while at Station 20, particulate concentrations were higher than discharge concentrations during the low-flow generation cycle. The return of water pooled in Oseligee Creek, located immediately upstream from Station 20, during peaking generation, and associated material transport may be the source of observed increases in iron concentrations. Total iron concentrations decreased to discharge concentrations (0.5 mg/L) coincident with the arrival of peaking-generation water. As observed for manganese, total and particulate iron concentrations increased to above discharge concentrations at Station 40 with the rising limb of the peaking-generation hydrograph. Increased transport of resuspended material could account for the increased concentrations.

Changes in water quality during near-steady state conditions provided additional information concerning physicochemical processes in the discharge. Temperature and DO values increased from 26.5 to 27.2 °C and from 2.3 to 3.1 mg/L, respectively, during the 3.5-hr period of travel from the dam to Station 50 (Figure 6). Increased DO concentrations suggest that some reaeration occurred during peaking generation. Both manganese and iron concentrations decreased (Figure 6). Between Stations 10 and 50, total iron concentrations decreased from approximately 0.55 to 0.40 mg/L and dissolved iron from 0.29 to 0.09 mg/L. Total manganese, primarily in a dissolved form, decreased from 0.37 to 0.26 mg/L. Assuming first-order loss reactions, the loss rate for dissolved iron was $K = 0.014/\text{min}$; the rate for manganese was $K = 0.002/\text{min}$ (Figure 7). These rates were comparable to those reported by Dortch, Hamlin-Tillman, and Bunch (1991). Although removal of both metals was observed in the tailwater, removal of iron occurred at a much faster rate than for manganese.

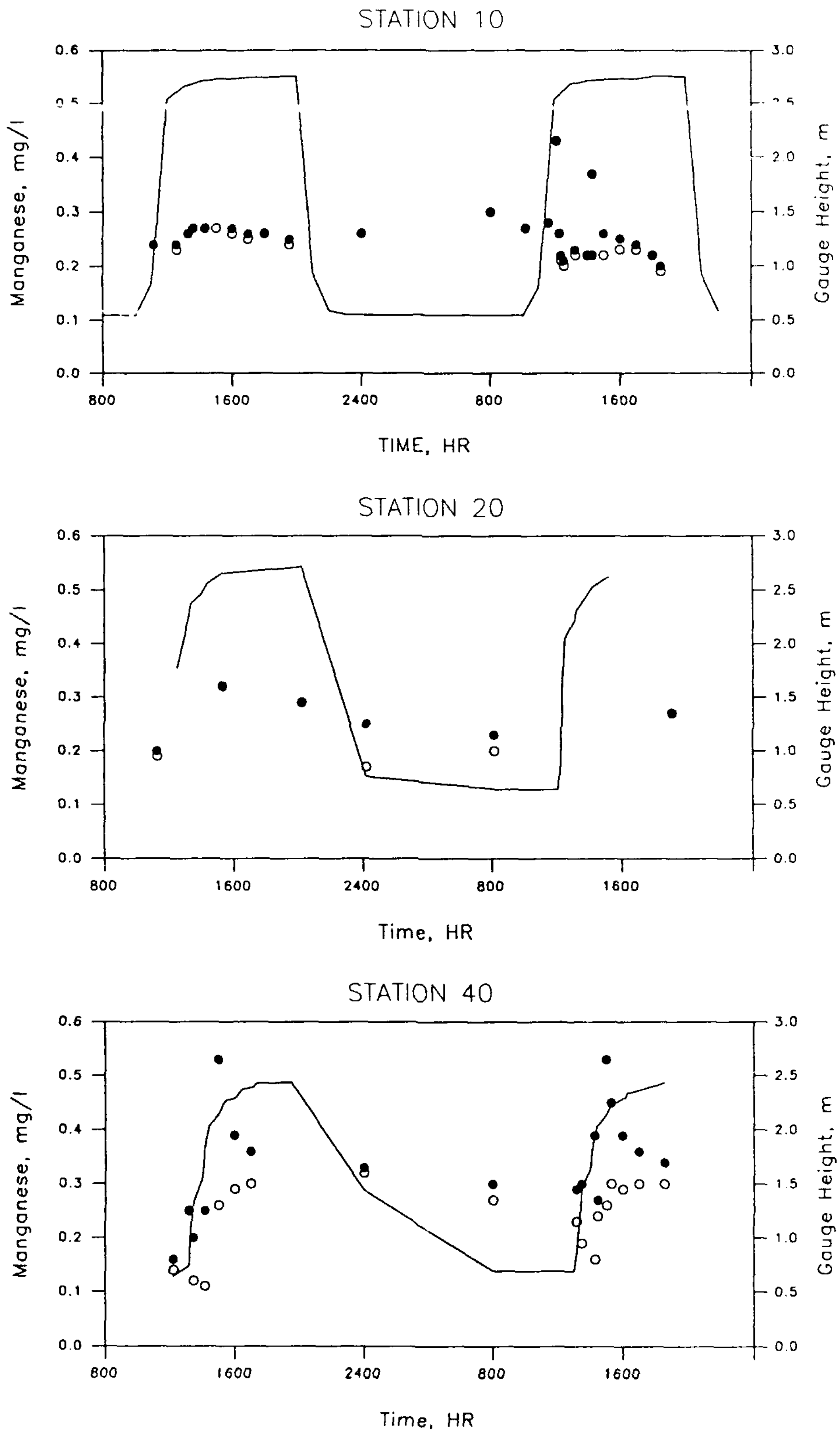

Figure 4. Manganese concentrations at Stations 10, 20, and 40

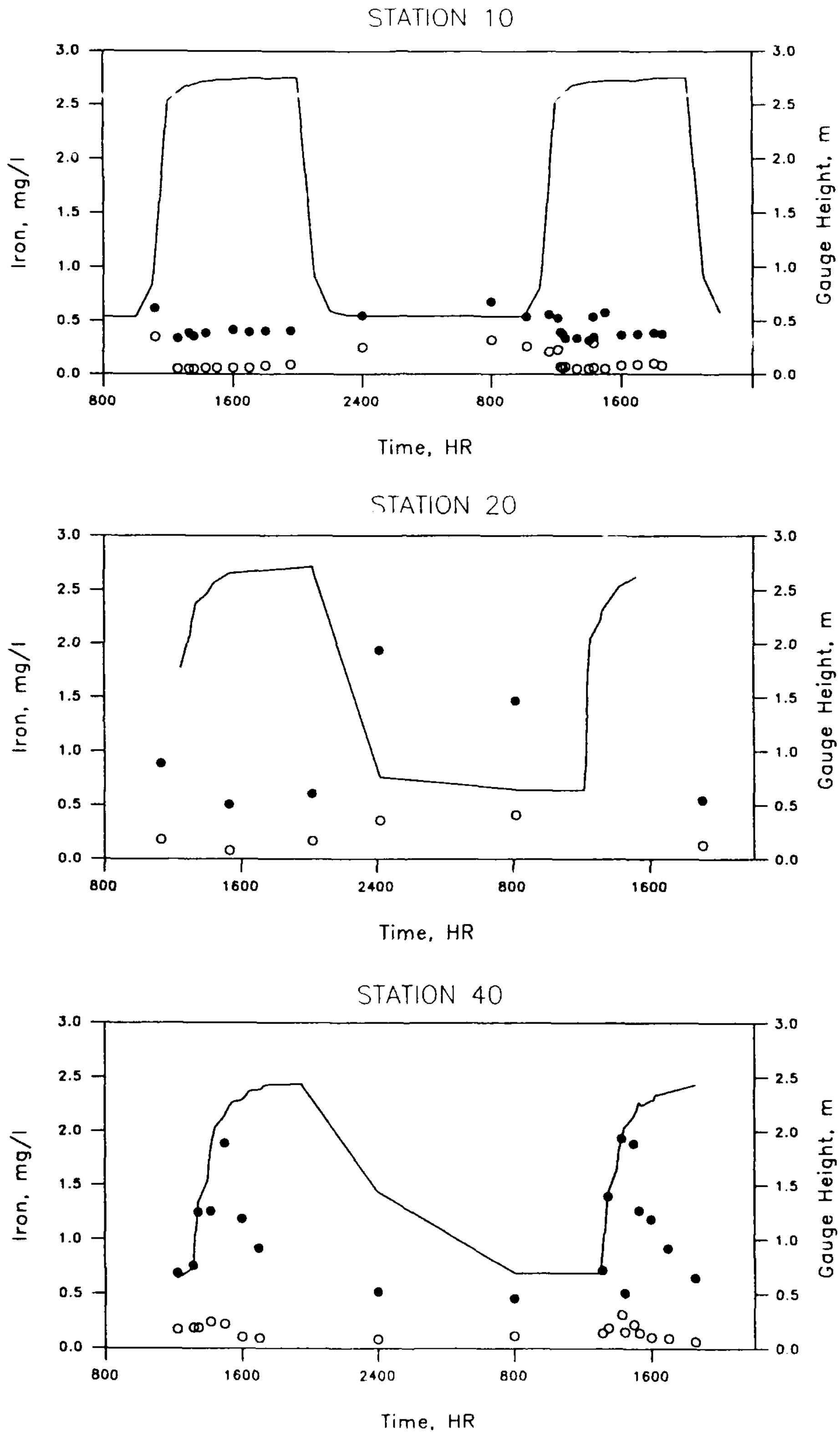

Figure 5. Iron concentrations at Stations 10, 20, and 40

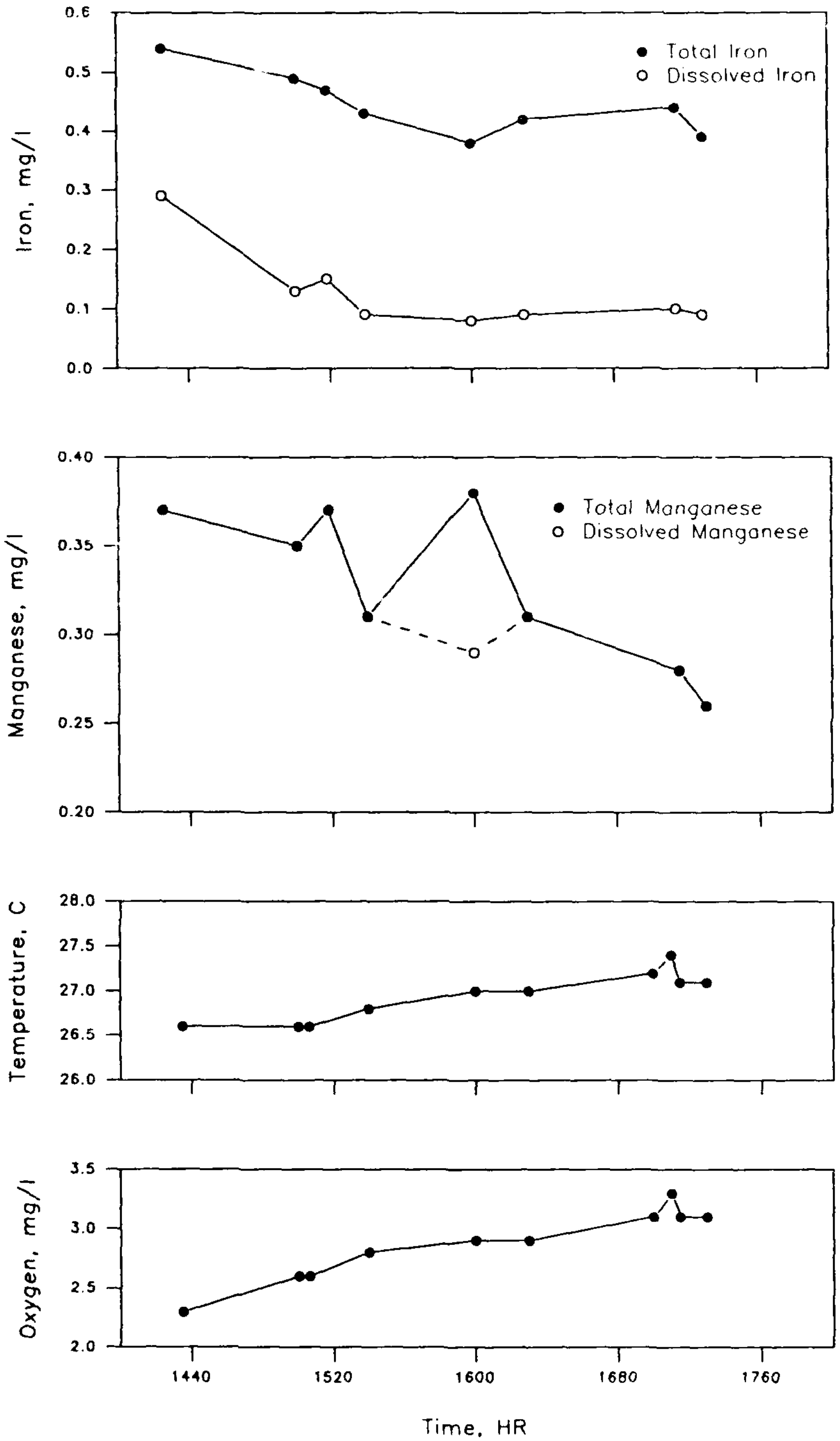

Figure 6. Response of physicochemical constituents during the time-of-travel study

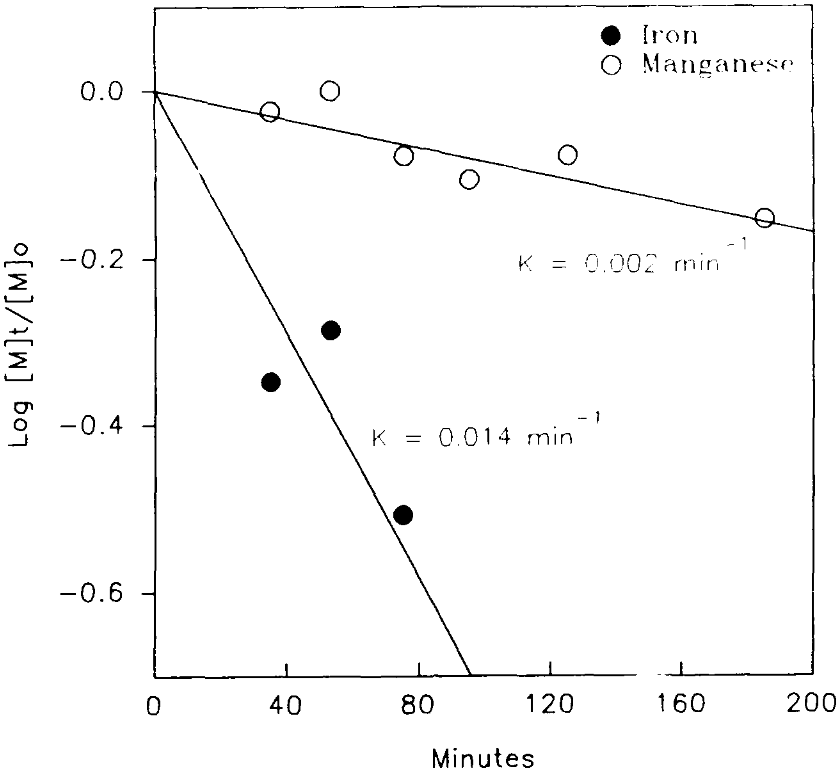

Figure 7. Loss rates for iron and manganese

Conclusions

Many factors influence the quality of water released from West Point Dam. Iron and manganese concentrations within the tailwater during late July were found to be influenced by (a) hydrograph dynamics, including transport, bottom scour, and resuspension; (b) local impacts, such as contributions from inflow streams; and (c) "loss reactions," a combination of factors affecting concentrations of dissolved metals, including but not limited to sedimentation, adsorption reactions, and oxidation-reduction processes.

Acknowledgments

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J. Strom Thurmond (Clarks Hill) Lake: Historical Perspective Based on Temperature and Dissolved Oxygen

by
Joe H. Carroll[1] and John J. Hains[1]

Introduction

Heterogeneities in reservoir water quality are dependent on a variety of related factors including morphometry, hydrology, external and internal loading of dissolved and suspended material, and climate. The understanding of long-term temporal variability in reservoir water quality requires an evaluation of changes in precipitation, inflow quality and quantity, operational strategies, and watershed use, perturbations and characteristics, as well as an appropriate and consistent long-term water quality monitoring of inflow, outflow, and in-pool stations (Kennedy, Thornton, and Ford 1985; Kimmel and Groeger 1986; Gaugush 1987). The impact of hydrology on spatial patterns in reservoir quality has been well documented (Love 1978; Kennedy, Thornton, and Gunkel 1982; Knowlton and Jones 1989). Since water throughput in a reservoir may vary greatly from one year to the next, sampling programs for evaluating trends in historical or long-term data must be able to account for this source of variability in determining the relevance or impact of water quality conditions.

The efforts to describe long-term historical trends in dissolved oxygen (DO) concentration and temperature patterns in a large, southeastern reservoir using a variety of sources for water quality, climate, and hydrology data are reported here.

Site Description

The Savannah River basin is long and relatively narrow, with its long axis lying in a northwest-southeast direction. The maximum length of the basin is nearly 402 km; maximum width is approximately 113 km. The total area of the basin is 27,400 sq km. The Savannah River originates on the southern slope of the Blue Ridge Mountains in North Carolina and flows in a southeasterly direction through the Piedmont Plateau and Coastal Plain along the boundary between Georgia and North and South Carolina. The upstream 160-km section of the Savannah River basin (Figure 1) includes three U.S. Army Corps of Engineers impoundments and three Duke Power Company hydropower projects.

J. Strom Thurmond Lake is a 300-sq km Corps of Engineers reservoir located on the Savannah River. Congressionally authorized reservoir purposes include hydroelectric power generation and water control for flood protection and navigation. The reservoir is also an important recreation resource. Originally known as Clarks Hill Dam, Thurmond Dam was constructed approximately 20 km upstream of Augusta, GA, and filled by 1954. It was the first major impoundment on the Savannah River. Thurmond is a long (63-km) dendritic

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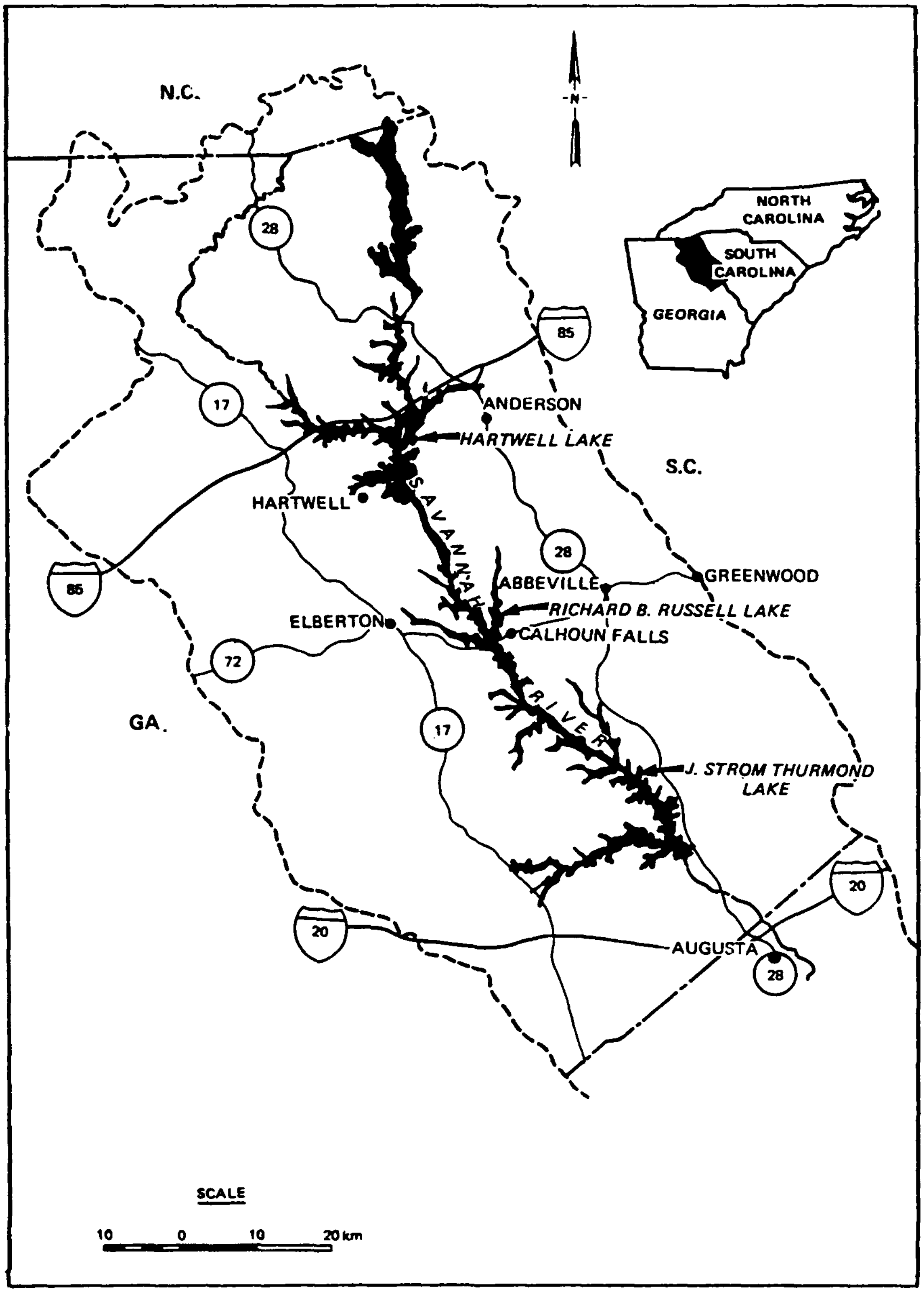

Figure 1. Upper Savannah River watershed

(shoreline ration of 32.0) impoundment with one large secondary embayment, the Georgia Little River arm, which extends 50 km west of the dam. Mean and maximum depths are 11 and 45 m, respectively. At maximum power pool elevation (100.6 m National Geodetic Vertical Datum, NGVD), the reservoir has a volume of 3.023 billion cu m. Average discharge from the reservoir is 240 cu m/sec, and the theoretical hydraulic retention time at maximum power pool is 144 days. More than 50 percent of the inflow to Thurmond reservoir is from the Savannah River; the second highest inflow to the lake, at approximately 20 percent, is from the Georgia Broad River.

Thurmond Lake is a warm monomictic reservoir exhibiting strong thermal stratification, low nutrient and dissolved solids concentrations, low alkalinity, and a low buffering capacity. Algal biomass is generally low, and algal blooms are infrequent. During periods of stratification (April-October), hypolimnetic anoxia develops from the dam to near midpool, with concomitant increases of iron and manganese concentrations in the anaerobic waters.

Since the impoundment of Thurmond, two Corps of Engineers impoundments and three Duke Power Company hydropower projects have been added upstream. The Georgia Power Company has several smaller projects on the upstream Tugaloo River arm of the Savannah River basin. The first of the major projects added was Hartwell Dam, which was completed in 1963; then, Keowee and Jocassee Dams (Duke Power), in 1971-1973. The next Corps of Engineers project was Richard B. Russell Dam, located in the headwaters of Thurmond and completed and filled by 1985. Bad Creek Dam is the most recent Savannah River basin project, completed in 1991 by Duke Power.

Methods

Database

Daily rainfall and median air temperature data for the period 1930 through 1989 were obtained from Earthinfo, Inc. (climate data, NCDC Daily Observations) for Calhoun Falls, SC (Weather Service Forecast Service station ID number 1277). The climate data for the years 1990 and 1991 were provided directly by personnel operating the Calhoun Falls weather station. This station is located 10 km northeast of Richard B. Russell Dam.

Hydrology data for J. Strom Thurmond, Richard B. Russell, and Hartwell reservoirs were obtained from the U.S. Army Engineer District (USAED), Savannah.

Dissolved oxygen and temperature data collected prior to 1983 were obtained from numerous water quality data reports and water quality studies (Ford 1962; U.S. Department of Health, Education and Welfare 1964; USAED, Savannah 1969, 1970, 1971, 1972, 1976; Water and Air Research, Inc. 1981; Applied Biology, Inc. 1982; James H. Carr and Associates, Inc. 1982). The data collection techniques have changed considerably over the course of the study from the use of temperature thermistors and manual Winkler DO titration to multiparameter in situ profiling equipment. Since 1983, all in-lake water quality sampling has employed Hydrolab Surveyor II and Surveyor III instruments (Hydrolab Corporation, Austin, TX). In situ profiles were obtained at selected stations at intervals necessary to describe depth-related differences for the 1983 to 1991 data. Since 1985, release water temperature and DO concentration have been monitored hourly using Schneider Instruments water quality monitors (Cincinnati, OH) below each of the Corps of Engineers dams.

Station and data selection

An effort was made to select two lake stations for J. Strom Thurmond Lake, which had been sampled by the majority of the historical studies in order to provide a continuum of appropriate data throughout the period of study. Sites selected were Station 20, which is representative of the near-dam or forebay area, and Station 30, which is located midway upstream in the main Savannah River thalweg (Figure 2). Lake data were minimal in the earlier years. Only temperature data were collected prior to 1966 for the two stations. September data were used to provide information from comparable seasons for the yearly evaluations. Thurmond Lake is always stratified during this month of the year, exhibiting large vertical gradients in both temperature and DO.

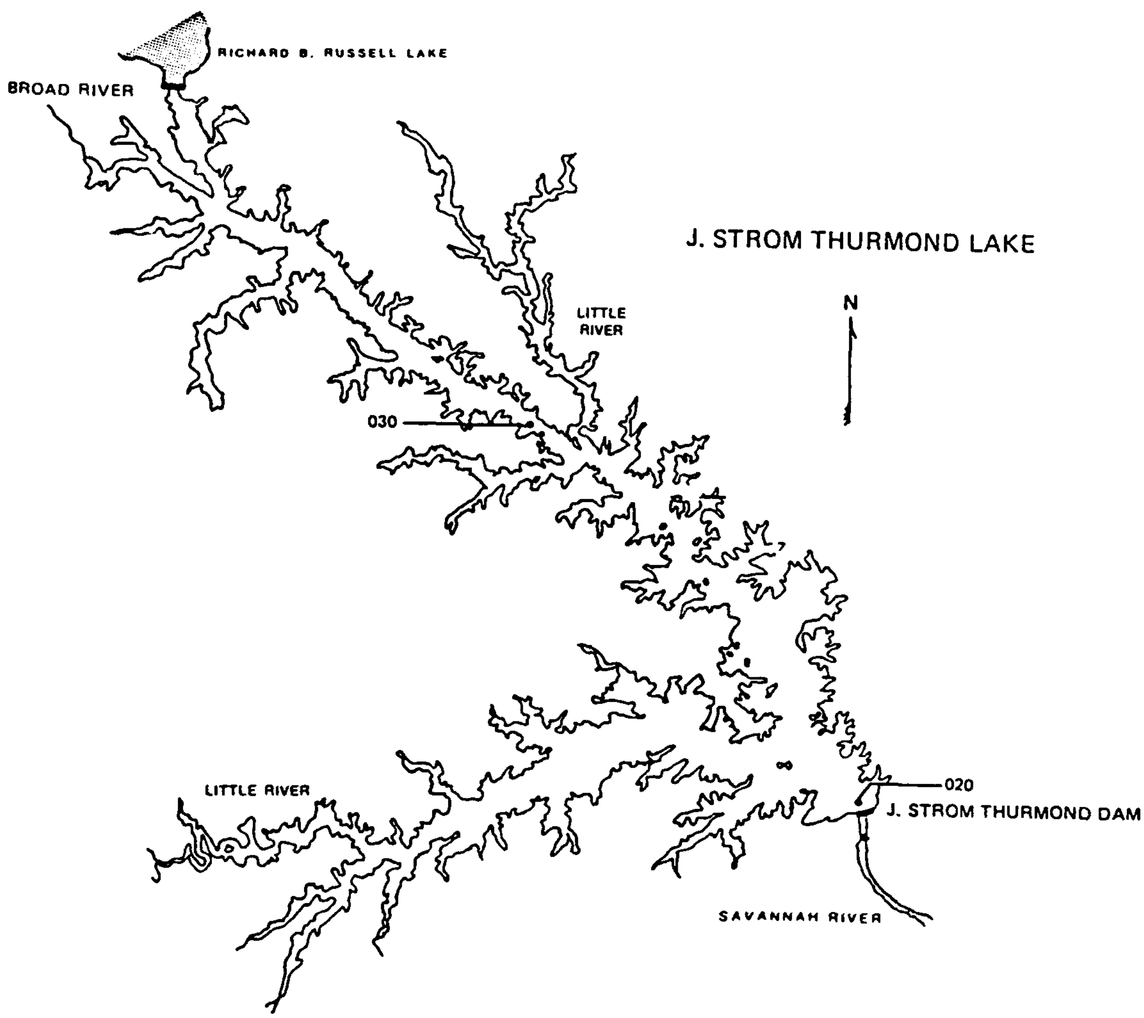

Figure 2. Sampling station locations, J. Strom Thurmond Lake

Climate and hydrology data to be used in this presentation are limited to the period 1961 through 1991, which coincides with the bulk of the water quality data reviewed.

Results

Rainfall accumulation data for the period 1962-1991 display wide variations both monthly and yearly (Figure 3). Monthly accumulations ranged from a minimum of 0 cm to a maximum of 30 cm, with a monthly mean of 9.7 cm. Although annual cycles in monthly rainfall were discernible with less accumulation occurring in the summer months, this pattern was often the exception. The annual amount of precipitation ranged from 80 to 180 cm, with a mean of 116 cm. The 3 years with the least rainfall occurred consecutively in 1986, 1987, and 1988, whereas years with above-average precipitation were scattered somewhat randomly throughout the period 1961-1991.

Air temperatures varied little annually throughout the study period except for normal seasonal patterns (Figure 3). The average annual median was 16 °C with a range in median temperatures from 0 to 28 °C.

J. Strom Thurmond lake elevations generally followed rule curve schedules throughout the study period, with a fall-winter drawdown and a return to full-power pool elevations during the wet winter-spring months (Figure 4). Exceptions occurred during 1986 and again in late 1987 through early 1989, coincidental with the three minimum rainfall years. The surface elevation range for the period was from 95 to 101 m NGVD.

Monthly mean discharge data for both J. Strom Thurmond and Hartwell Lakes (Figure 4) varied greatly as did the monthly rainfall accumulation. Average monthly mean discharges for Thurmond and Hartwell were 231 and 113 cu m/sec, respectively. As with Thurmond surface elevation, the minimum mean discharges for the two lakes coincided with the minimum rainfall years 1986 to early 1989. Monthly mean discharge from Thurmond dam ranged from 100 to 1,077 cu m/sec whereas Hartwell dam discharged between 40 and 440 cu m/sec. Hartwell releases accounted for approximately 50 percent of the water being discharged from Thurmond, while direct runoff and secondary tributary inflow to Richard B. Russell Lake accounted for less than 10 percent of Thurmond discharges. The majority of the remaining release water from Thurmond corresponds to inflows from Broad River and Georgia Little River.

As evident in Figure 4, the completion and filling dates for Hartwell, Keowee, Jocassee, and Richard B. Russell Lakes had no distinct impact on Thurmond hydrology. With the completion and operation of Richard B. Russell Dam, however, significant changes in water quality characteristics of summer inflows to Thurmond are apparent (Table 1). For the years prior to the completion of Richard B. Russell, the summer monthly temperatures averaged 20.2 °C, and DO averaged 8.4 mg/L. During 1984 the Russell Dam tainter or surface gates were used to release water and resulted in a mean summer outflow temperature of 18.5 °C and a mean oxygen concentration of 9.6 mg/L. Depending on lake elevations, the tainter gates withdrew from the surface or deeper into the colder waters. This resulted in a 2 °C cooling of the waters flowing into Thurmond for that year. Oxygen levels increased to near-saturation for 1984 as the result of the extremely turbulent tainter releases. Since the beginning of 1985, Russell Dam has withdrawn water through the penstocks into the powerhouse, which results in considerably less turbulence and withdraws water from much colder depths that contain less oxygen. This has resulted in even more cooling of the water entering Thurmond, down to a summer average of 17.0 °C. Post-1985 average summer DO concentrations of the Thurmond inflows have decreased considerably to 6.0 mg/L.

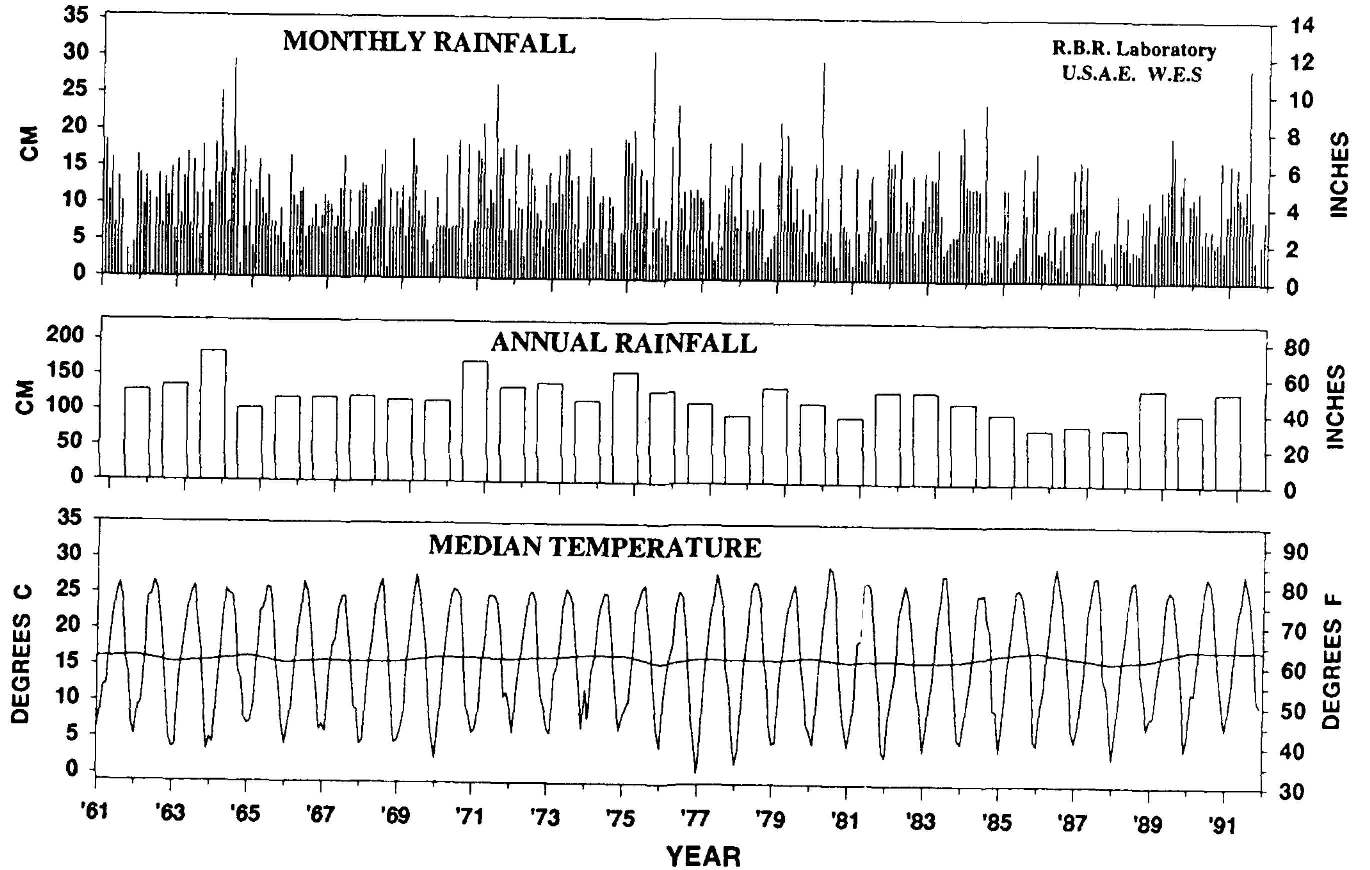

Figure 3. Climate data for Calhoun Falls, SC, 1961-1991

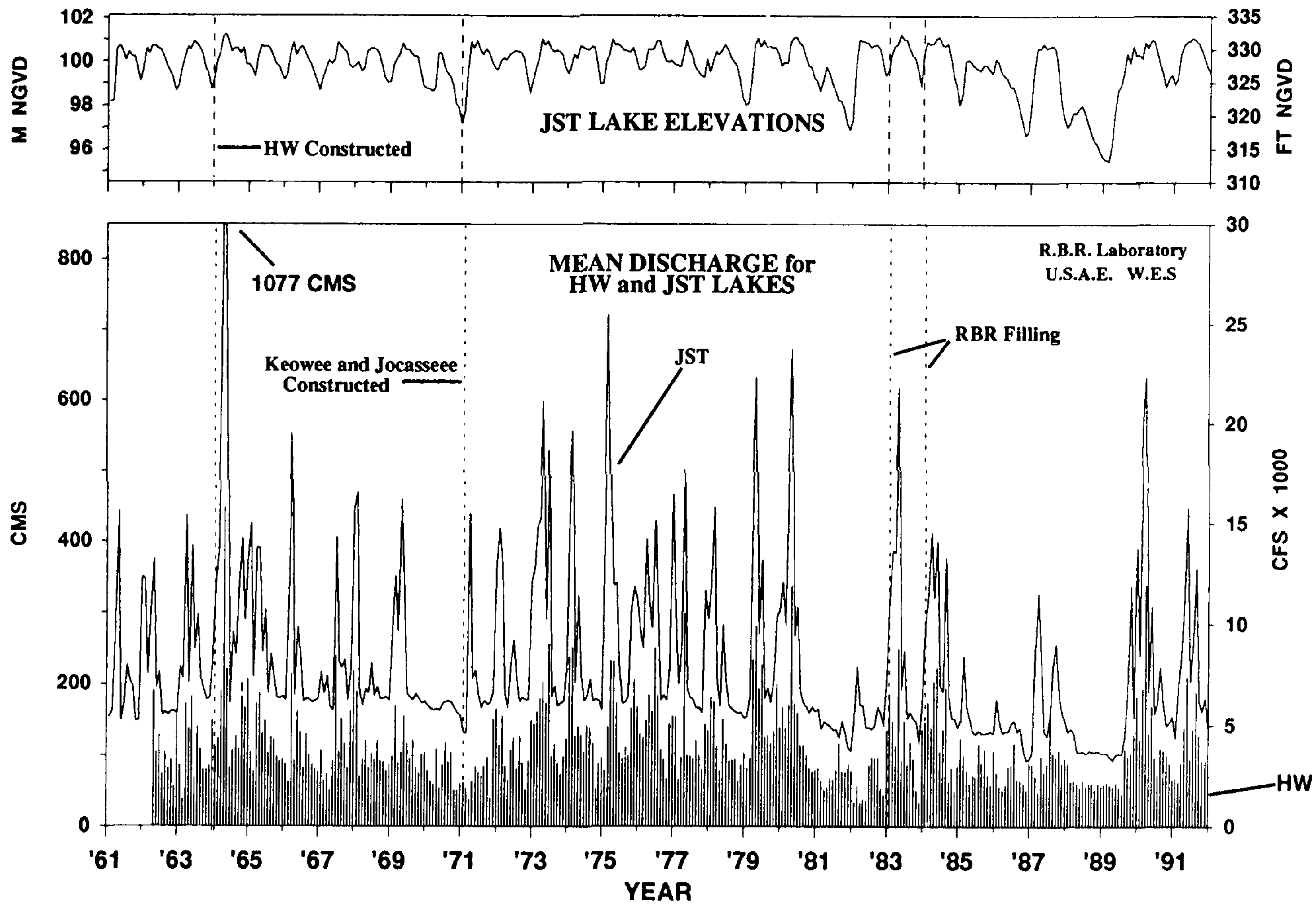

Figure 4. Reservoir hydrology for J. Strom Thurmond and Hartwell Lakes, 1961-1991

Table 1
Summer Month Mean Values for Temperature and Dissolved Oxygen, Savannah River Inflows to J. Strom Thurmond Lake

Year	Temperature, °C	DO, mg/L
1967	19.2	8.5
1968	20.7	8.0
1969	20.0	8.6
1970	20.9	9.5
1972	20.5	7.7
1973	20.1	7.7
1981	20.0	8.7
1984	18.5	9.6
1986	17.1	6.2
1987	15.7	6.3
1988	15.0	6.0
1989	16.7	5.9
1990	18.2	4.5
1991	19.1	6.9

Summer thermal patterns in Thurmond Lake have changed significantly since impoundment. During earlier years of impoundment the forebay epilimnion ranged from 12 to 18 m deep (Figure 5). Following impoundment of Richard B. Russell Lake and normal hydropower operation of the dam in 1985, the Thurmond forebay epilimnion has remained between 8 and 10 m deep with a much larger hypolimnetic volume than existed prior to Russell Lake impoundment. During 1984 tainter gate releases, thermal patterns were similar to later years; however, warmer water extended deeper to the water column.

Temporal DO concentration patterns for the Thurmond midlake region (Figure 6) indicate a decrease in hypolimnetic oxygen content following normal hydropower operation at Richard B. Russell Dam in 1985. However, this comparison uses only one year, 1984, preceding Russell hydropower operation.

September DO profiles in the Thurmond forebay indicate a major shift in hypolimnetic oxygen content and pattern (Figure 7). Many of the years prior to and including 1984 were characterized by hypolimnetic maxima peaks in DO, often with as much as 2.5 mg/L more DO than above and below the peak. The years following 1984 and the initiation of hydropower operation of Russell Dam were dominated by near-anoxic hypolimnetic waters in the forebay area of Thurmond lake. For the post-1984 period, a hypolimnetic DO maxima was present only during September 1991, a year that was characterized by high monthly rainfall and water discharges throughout the spring and entire summer months. Another noticeable difference in Thurmond DO profiles following 1984, like temperature data, is a shallower surface mixed layer or epilimnion. This layer was 10 m or less following 1984, whereas the earlier September data showed it to be 10 m or greater.

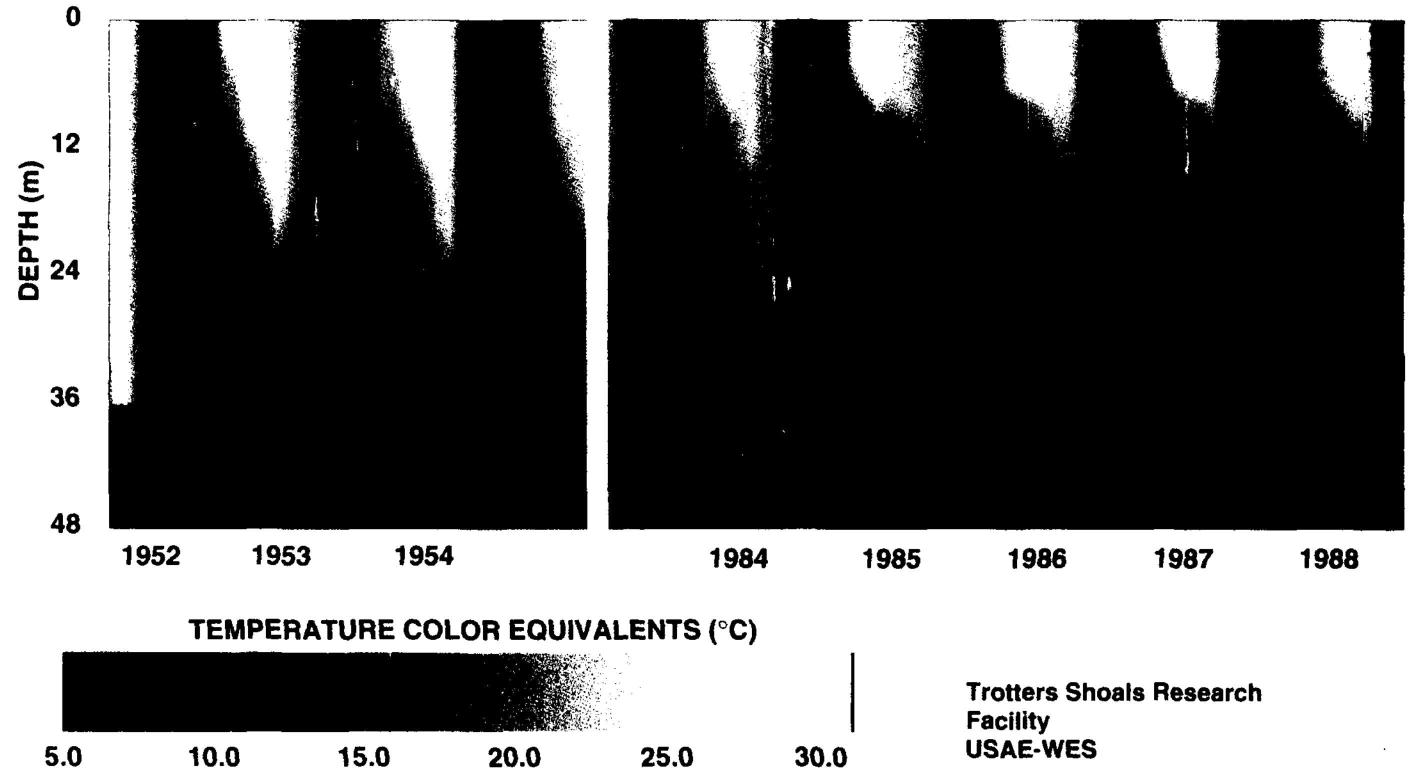

Figure 5. Temporal patterns in thermal structure for selected years, J. Strom Thurmond Lake

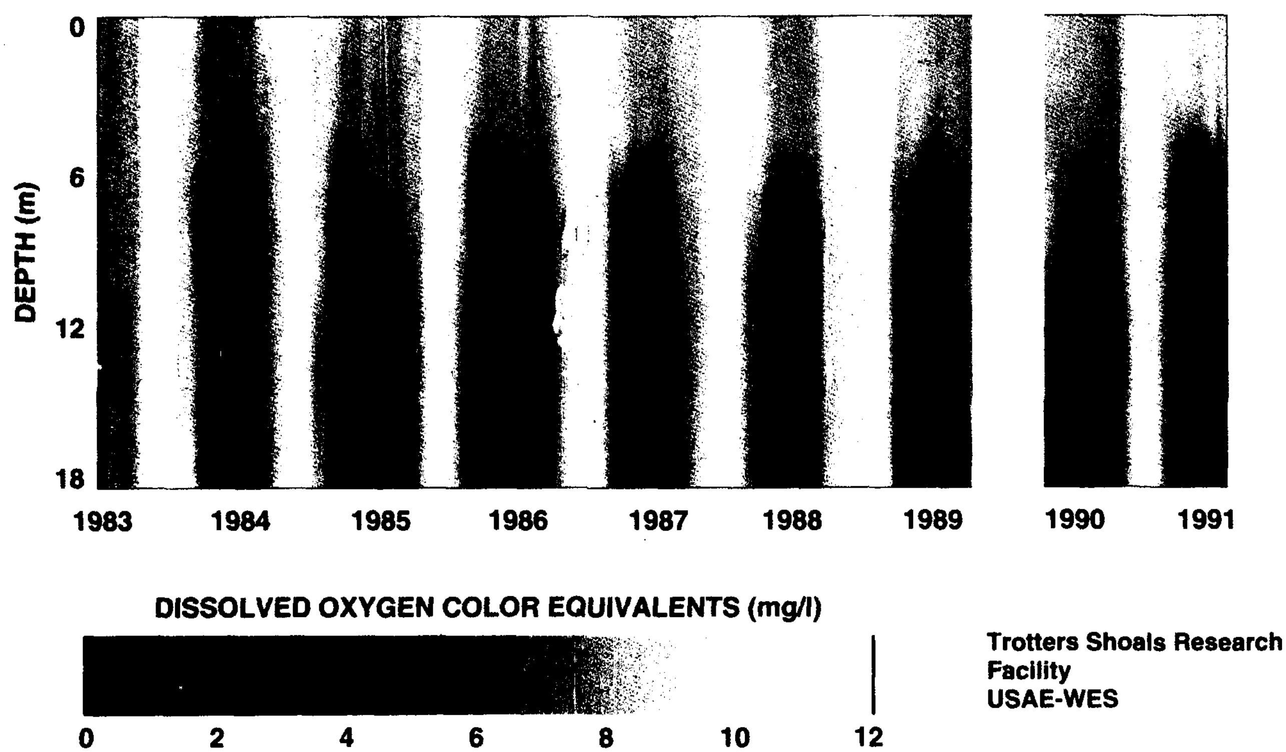

Figure 6. Midlake DO patterns, 1983-1991, J. Strom Thurmond

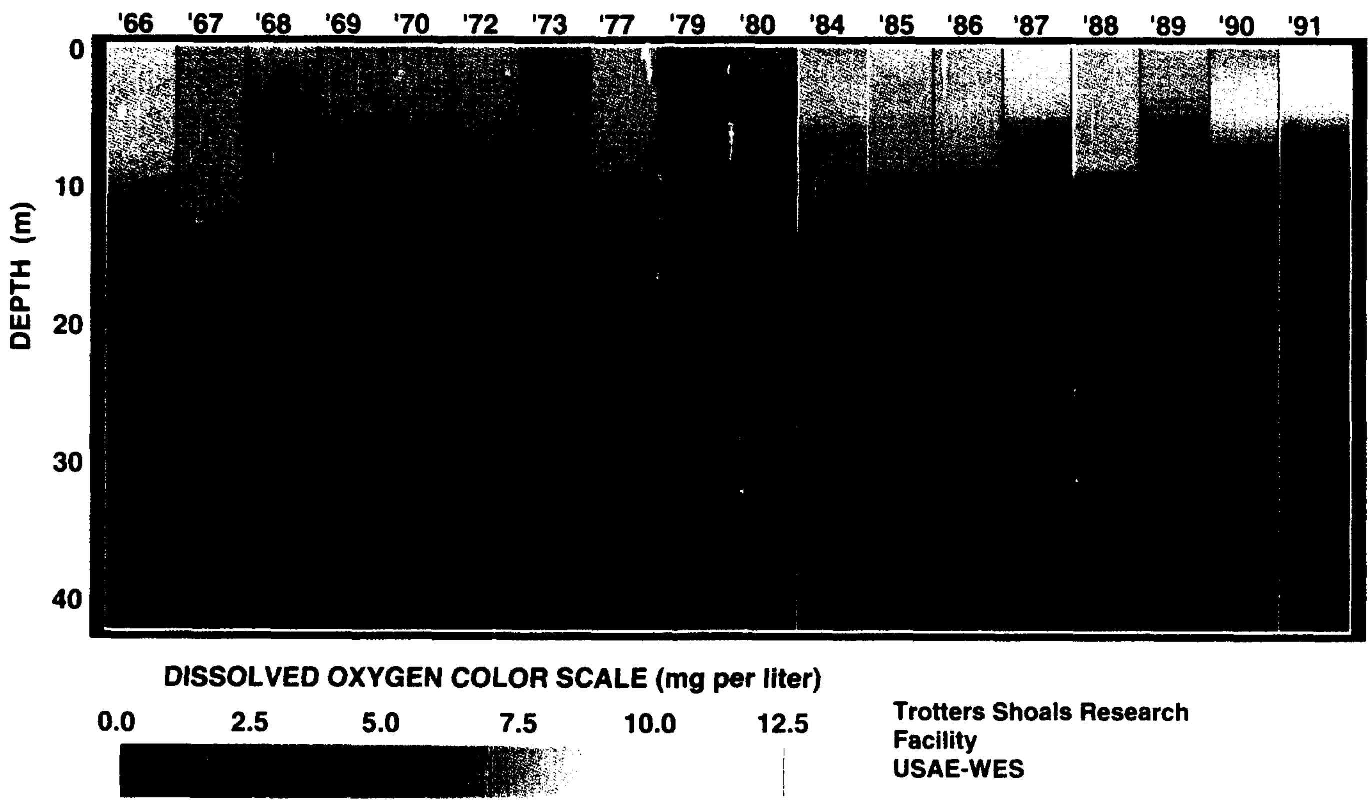

Figure 7. Forebay DO profiles for September (1966-1991), J. Strom Thurmond Lake

Discussion

Often, temporal trends in reservoir water quality are considered a rather consistent and progressive development such as a gradual change in trophic state over the years resulting from increased loading. Evidence from J. Strom Thurmond shows that this is not necessarily the case with DO and temperature patterns (Figures 5-7). Prior to the construction of Richard B. Russell on the Savannah River headwaters of Thurmond, oxygen patterns were generally similar from year to year with occasional outliers. After Richard B. Russell Dam began hydropower operation in 1985, abrupt changes are apparent in both oxygen and temperature. Hypolimnetic waters became consistently larger in volume, oxygen concentrations decreased overall with hypolimnetic maxima occurring only in September 1991, and overall water column temperatures decreased with a somewhat shallower epilimnion. Hydrology for 1991 was quite different from other post-1984 years because of the greater discharges and rainfall throughout the summer months in the Savannah River basin. This may explain the higher hypolimnetic oxygen content in Thurmond Lake for that year.

No relationships were recognizable between hydrology or climate data and the water quality data except for 1991, as mentioned above. The most severe drought years for the period of study occurred during 1986 to 1988, with no noticeable impacts on water quality patterns in Thurmond.

Conclusion

In conducting water quality evaluations or studies on reservoirs, it is important to consider basin and time-scale attributes. Any major change in basin characteristics such as flow quality or quantity can impact both temporal and spatial water quality patterns or trends. The importance of evaluating water quality information from a river system approach cannot be overstated. The addition of a major impoundment upstream of an existing structure such as J. Strom Thurmond Dam has the potential to alter longitudinal and vertical quality patterns throughout the pool.

Review of temperature and DO data provides evidence that important changes in water quality have occurred since impoundment of J. Strom Thurmond Lake in 1954. Most notable are changes associated with the construction and operation of Richard B. Russell Dam and Lake immediately upstream. Prior to 1984, when Richard B. Russell Dam was completed, Savannah River discharges into the lake were regulated by the operation of Hartwell Dam. Since 1984, Savannah River discharges in the lake have been controlled by the operation of Richard B. Russell Dam. Coincident with this change there has been a decline in hypolimnetic temperatures and DO concentration.

Year-to-year fluctuations in water quality and quantity can be as great as the difference over the life of the project. Studies that do not account for annual variability in water quality should be used carefully in making water quality management decisions for reservoirs.

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Fish Spill and Dissolved Gas Saturation at Columbia and Snake River Dams

by
A. Rudder Turner, Jr.[1]

Introduction

The Corps of Engineers implements substantial water management and project operations measures each year, and conducts research to enhance salmonid survival at its dams on the Columbia and Snake Rivers in the Pacific Northwest region. These actions include (a) operating adult and juvenile fish passage facilities, (b) collecting and transporting juvenile fish past main stem dams, (c) releasing water from storage reservoirs to flush fish downstream, (d) providing voluntary spill to allow spillway passage, (e) monitoring fish migrations and water quality, (f) operating main stem reservoirs to reduce fish travel time, and (g) conducting fisheries research to improve project passage conditions and increase survival. Fish spill is provided based on the assumption that fish passage mortality is lower for spillway than turbine passage routes. Fish spill management must consider water quality impacts, as total dissolved gas concentrations can approach lethal levels for fish downstream of spillways if not carefully controlled.

The current fish spill program has been in effect since 1989, when the Northwest Power Planning Council (NPPC) amended its Fish and Wildlife Program to incorporate provisions of a regional 10-year spill agreement (NPPC 1989, 1991) that affects several Corps projects (Figure 1). In addition, substantial spill occurs at Bonneville Dam as a result of restrictions on second powerhouse operations during the juvenile fish passage season. The resulting high spill levels have important water quality management implications in terms of dissolved gas control. This issue becomes more significant in light of recent listings of several sockeye and chinook salmon stocks as endangered and threatened under provisions of the Endangered Species Act (ESA).

Fish Spill Program

Background

The Corps began spilling for juvenile fish passage at its main stem projects in 1977, a year of severe regional drought. From 1977 to 1984, projected flow conditions were reviewed in late winter, in coordination with the fishery agencies, Indian tribes, and utilities, to determine the extent of the year's spill program. Project spill amounts and durations were provided at the discretion of the Corps based on nightly fish passage monitoring. The first NPPC Program, adopted in 1982, called for the Corps, in consultation with the fisheries agencies and tribes, to implement a spill program that would achieve fish survival comparable to the best available bypass system. However, agreement was never reached among the parties involved over spill levels necessary to meet the Program's survival goals. The

[1] U.S. Army Engineer Division, North Pacific; Portland, OR.

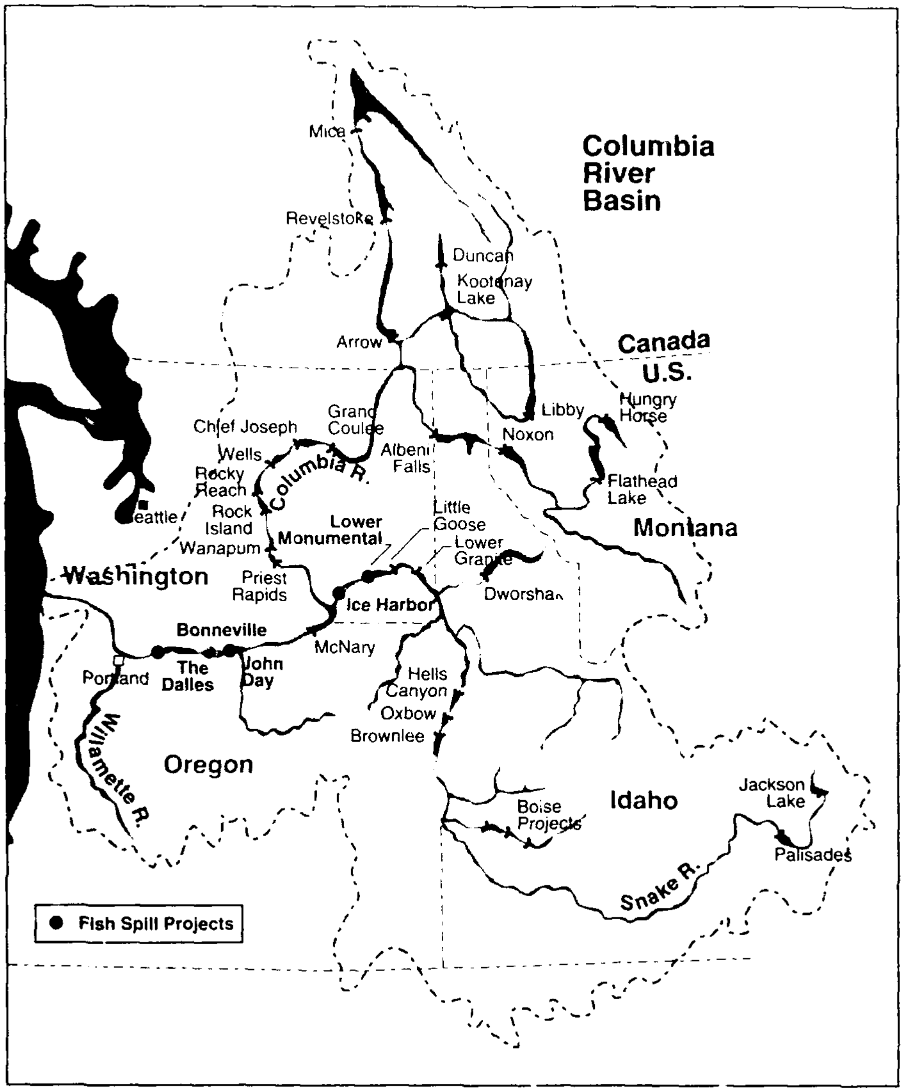


Figure 1. Major Columbia River basin main stem and storage dams

fisheries agencies disputed the Corps' project survival estimations and would not support the final Juvenile Fish Passage Plan (JFPP) annually provided to NPPC.

Beginning in 1987, the Corps settled on a JFPP that called for spill at John Day and Lower Monumental Dams to meet or exceed the Program's 90-percent dam passage survival goal. Hydroacoustic equipment and a monitoring team were deployed at the two dams to develop a daily project fish passage estimate. Spill was provided when a threshold passage level was reached. Also in 1987, the fisheries agencies, tribes, utilities, and NPPC began negotiations on a spill agreement. This agreement resulted in preset spring and summer spill levels and duration at The Dalles, John Day (summer only), Ice Harbor, and Lower Monumental Dams until adequate bypasses were installed, or for a period of 10 years. The Corps did not participate in negotiations, nor did the Corps sign the agreement. In 1989, shortly after the parties signed the agreement, NPPC amended its Program to reflect the agreement's provisions, including the 10-year period. The Corps agreed to implement the Council's spill program in 1989 and has continued to do so on a year-to-year basis until the present time.

Spill amendments and Bonneville Dam operations

The spill amendments call for nightly spill at The Dalles during the period May 1-August 22, at a level of 10 percent of daily average discharge through June 6, and 5 percent of daily discharge until the end of the spill season. Spill normally occurs 8 hr nightly (2000-0400 hr); as the result of compression of spill hours, levels are typically 30 percent of instantaneous in the spring and 15 percent of instantaneous in the summer seasons.

At John Day, spill is provided nightly from June 7 to August 22, for 10 hr (2000-0600 hr), at a level of 20 percent of instantaneous project discharge. Spill at Ice Harbor occurs between April 15 and July 22, for 12 hr nightly (1800-0600 hr), at a level of 25 percent of instantaneous project discharge. Lower Monumental provides spill on the same dates and hours as Ice Harbor, at a 70 percent instantaneous level.

These dates, times, and amounts are the results of agency agreements and, as such, incorporate both biological rationale and political/legal compromises into the final provisions. For 1992, additional summer spill is under consideration at Ice Harbor and Lower Monumental for ESA-listed stocks.

Spill for fish occurs at Bonneville Dam as a consequence of restrictions on second powerhouse operations during the juvenile fish migration season (U.S. Army Engineer Division (USAED), North Pacific 1992). The restrictions are in place because of poor juvenile fish guidance at the second powerhouse bypass facility. Columbia River flows peak during the fish passage season and usually exceed first powerhouse hydraulic capacity. Daily average spill levels are typically 30 to 40 percent of total project discharge during the fish passage season.

Spill for fish passage also occurs at mid-Columbia projects owned by Public Utility Districts (PUDs), based on Federal Energy Regulatory Commission license requirements. This daily spill keeps total dissolved gas (TDG) levels high downstream to McNary Dam. In addition, heavy spill at Canadian projects causes high TDG levels coming across the international boundary into Lake Roosevelt.

Dissolved Gas Monitoring

A large body of literature exists on nitrogen saturation effects on fish in the Columbia River basin (summarized in Ebel et al. 1975; Dawley 1986; and U.S. Environmental Protection Agency, USEPA 1986). Also, predictions of in-river fish survival have shown a decrease at higher river flows (Sims et al. 1983), in theory as the result of high dissolved gas levels from spill. Gas bubble disease in fish results from exposure to high dissolved nitrogen levels in water, with the severity of symptoms dependent on supersaturation level, exposure duration and depth, water temperature, and fish vitality. Problems associated with high gas levels resulted in flip lip installation at the spillways of several Corps projects in the 1970s and development of a TDG monitoring network to assist in spill management.

The TDG monitoring network has operated since 1979 and has been in its present configuration since 1986. Seventeen automatic stations transmit data hourly from March to October, by GOES satellite and teletype, to a system operations database in Portland. The network and database are managed by the North Pacific Division's Reservoir Control Center,

Fish and Water Quality Unit, with participation by the U.S. Bureau of Reclamation and mid-Columbia PUDs. The network monitors TDG and water temperature. Most stations are located in project forebays; however, gages are located downstream of Bonneville and Grand Coulee Dams. As Bonneville has regular spill during the fish passage season, comparison of records between the Bonneville project forebay and a gage at Warrendale, OR, about 6 miles downstream, shows the extent of gas saturation increase caused by project spill. Additional tailrace data were collected in 1991 by the USAED, Walla Walla, below Lower Monumental and Ice Harbor Dams to monitor seasonal fish spill, and below Lower Granite Dam in conjunction with a spill test where 100-percent spill occurred for 4 hr on June 1, 1991 (USAED, Walla Walla 1992).

Results

Spill discharges

Annual fish spill volumes under the current program, during 1989-1991, have ranged from 1.82 to 2.74 million acre-feet (MAF) at The Dalles, 1.09 to 1.67 MAF for summer spill at John Day, 1.01 to 1.21 MAF at Ice Harbor, and 3.22 to 4.10 MAF at Lower Monumental. Bonneville has spilled between 11.4 and 17.6 MAF annually since 1989, incidental to second powerhouse fisheries restrictions. These large spill volumes are in excess of involuntary or overgeneration spill, and system power generation has been impacted. Energy purchases have been required and, as a result, some power sales opportunities lost. Annual generation losses have ranged from 688,000 to 757,000 megawatt-hours (MW-hr) for the spill amendment projects and 575,000 to 860,000 MW-hr at Bonneville (Table 1). These losses have cost the power system an estimated $8.47 to $12.9 million at the spill amendment projects combined and $8.98 to $12.2 million at Bonneville each year. This spill has occurred in years of below-normal to normal Columbia River basin runoff volumes.

Juvenile fish survival

Hydroacoustic monitoring of juvenile fish passage occurred in 1989 at Lower Monumental and John Day Dams; estimates of fish passage, project passage survival, and fish saved by spill were made (USAED, North Pacific 1990) (see Table 2). Estimated project fish survival was enhanced by the 1989 spill program over what would have occurred if the Corps' JFPP criterion of previous years had been used to initiate spill. Survival rates were 90.2 to 93.8 percent, an increase of 0.4 to 4.4 percent over levels that would have been achieved with JFPP spill levels. Juveniles saved by spill increased by an estimated 35,000 to 134,000 fish (juvenile to adult survival is about 1 to 5 percent for most Columbia basin stocks). This represented a range of 32 to 63 fish saved per thousand acre-feet (KAF) of water volume spilled in 1989. Spilling each night also provided improved protection for depleted wild fish stocks over previous years, as these fish migrate downstream in smaller numbers over a longer period of time than do hatchery stocks.

Dissolved gas saturation

Total dissolved gas levels generally ranged from about 110 to 125 percent at monitored projects in 1991 during the late spring and early summer (reported in detail in USAED, North Pacific 1991). Runoff volumes were about normal in the lower Columbia, above normal in the upper Columbia, and below normal in the lower Snake River in 1991. Spill occurred at Bonneville between March 22 and August 20; daily mean TDG at Warrendale was 2 to

Table 1
Summary of Fish Spill Power Losses and Costs, 1989-1991, for Spill Amendment Projects and Bonneville Dam

	Spill Amendment Projects	Bonneville	Total	The Dalles, (Jan-Jul) Runoff Volume, MAF
		Power Loss, MW-hr		
1991	756,931	859,852	1,616,783	107.1
1990	733,755	724,166	1,457,921	99.7
1989	688,204	574,522	1,262,726	90.6

	Spill Amendment Projects	Bonneville	Total
		Cost	
1991	$8,470,756	$9,783,612	$18,254,368
1990	$12,920,817	$12,202,512	$25,123,329
1989	$12,484,298	$8,975,137	$21,459,435

Table 2
Estimated Project Survival and Spill Volumes for Observed 1989 Spill, Calculated Spill According to Corps JFPP Criteria, and Difference Between the Two[1]

	No. of Spill Days	Spill Volume KAF	Project Survival Percent	Days with 90% Survival	Fish Saved by Spill
			John Day Dam		
Summer					
1989 spill	71	1,086.6	90.2	46	37,248
JFPP spill	51	623.8	89.8	16	25,571
1989-JFPP	20	462.8	0.4	30	11,677
			Lower Monumental Dam		
Spring					
1989 spill	46	2,127.7	93.8	45	134,373
JFPP spill	47	1,525.1	91.2	32	92,497
1989-JFPP	-1	602.6	2.6	13	41,876
Summer					
1989 spill	42	1,091.3	91.7	42	35,429
JFPP spill	14	324.7	87.3	7	10,969
1989-JFPP	28	766.6	4.4	35	24,460

[1] Based on hydroacoustic monitoring of juvenile fish passage.

15 percent higher than at the Bonneville forebay during this period (Figure 2a). In September, with no spill at Bonneville, TDG levels were more similar. Daily TDG maximum was 127.3 percent at Warrendale, in late May. These levels prompted the Corps to impose a nighttime spill cap at Bonneville, resulting in levels returning to the 120-percent range at night, with daily averages between 110 and 120 percent. TDG levels at lower Snake and mid-Columbia projects generally followed spill trends as well (Figures 2b and 2c).

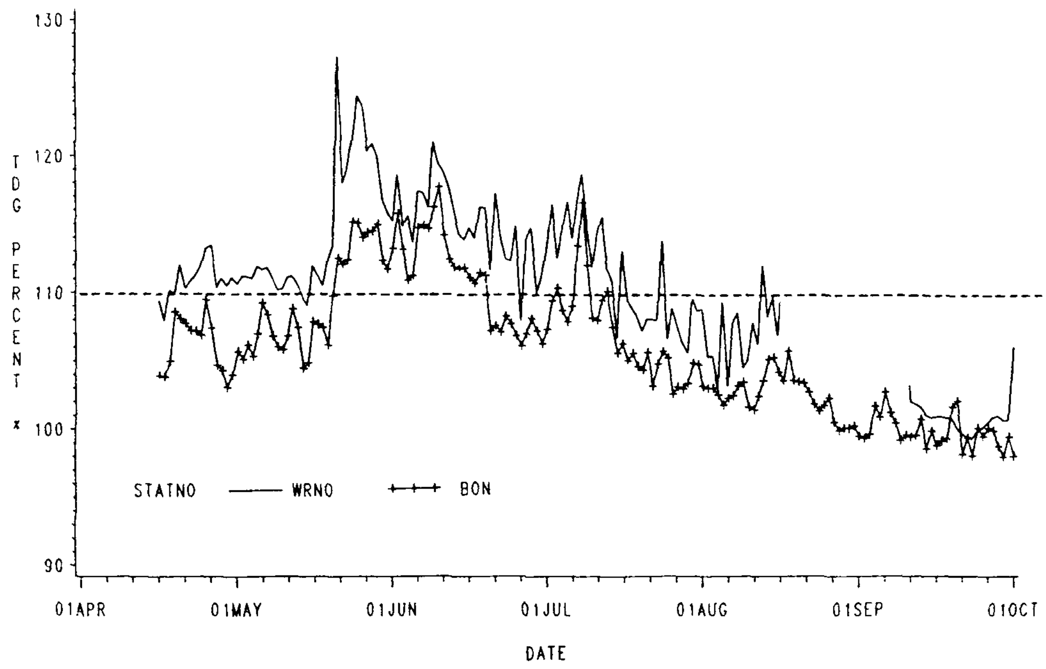
Daily mean dissolved gas levels have been compared between Bonneville and Warrendale for the last 6 years of monitoring records, 1986-1991 (Figure 3). Dissipation of dissolved gas between Bonneville and Warrendale varies with spill level at Bonneville, showing decreased dissipation ability as Bonneville spill percentage increases. Also, within a spill range, dissipation rate decreases slightly as TDG level at Bonneville increases. This indicates that, as the river becomes more supersaturated, there is less recovery with distance and time. A relationship with discharge and, thus, water travel time between Bonneville and Warrendale probably exists as well.

Monitoring of the 4-hr Lower Granite spill test showed TDG levels downstream of the spillway increasing from 101 to 126 percent within 30 min after initiating 100-percent spill at a discharge level of 105,000 cfs (USAED, Walla Walla 1992). TDG levels steadily increased to 137.9 percent at 3 hr, 15 min, into the test and were continuing to rise. The high TDG levels, combined with extreme hydraulic conditions in front of the powerhouse, led to the conclusion that poor passage conditions for adult migratory fish would result from high spill during reservoir drawdowns, should this be necessary. Tailrace TDG monitoring below Lower Monumental and Ice Harbor Dams showed elevated gas levels as well, coincident with fish spill. At Lower Monumental, with a 70-percent spill in effect, TDG exceeded 120 percent on 24 of 52 nights monitored and exceeded 130 percent on 7 nights. Supersaturation was less severe at Ice Harbor, where the spill level was 25 percent (Tom Miller, Walla Walla District, unpublished data).

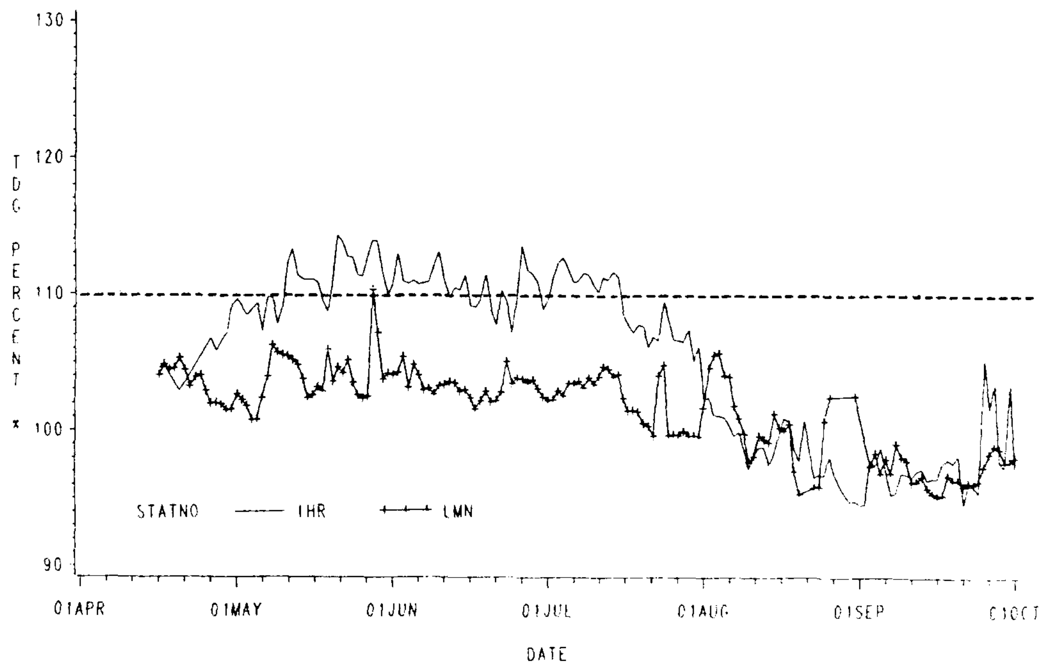
Discussion

Spill levels at the four spill amendment projects have approximately tripled over pre-1989 levels; combined with spill at Bonneville, the current program costs the region up to $25 million yearly from lost power generation. This spill also has contributed to dissolved gas levels, particularly in the spring, that chronically exceed Washington and Oregon Water Quality Standards as well as USEPA's "Gold Book" Water Quality Criteria (USEPA 1986). As a result, the Corps closely manages spill by issuing and revising seasonal spill priority lists that distribute overgeneration spill among projects. Objectives of spill prioritization are to distribute system spill more widely while providing fish passage benefits. Spill priorities are set through coordination with fisheries agencies, Indian tribes, and the Bonneville Power Administration.

Few incidences of gas bubble disease have been noted in the smolt monitoring program in recent years. However, chronic effects remain a concern. Total effects likely are underestimated in the current monitoring program. In-reservoir mortality may reduce numbers of affected fish prior to their arrival at the next dam downstream. This would be the first smolt monitoring point following fish exposure to high dissolved gas levels.

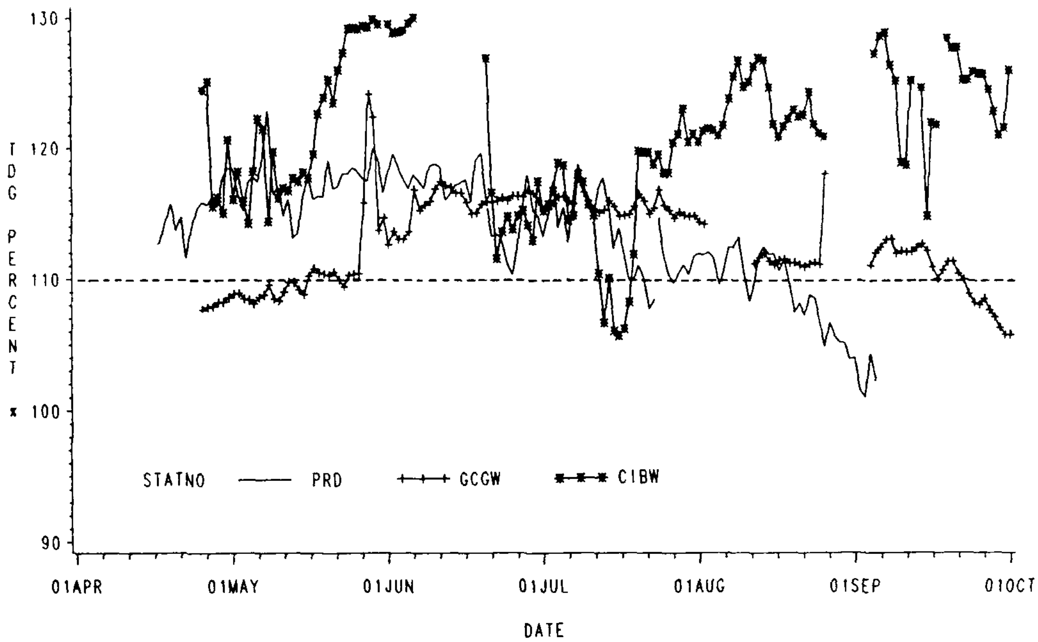


a. Bonneville and Warrendale stations, 1991



b. Ice Harbor and Lower Monumental stations, 1991

Figure 2. Daily mean of TDG saturation percent (Line at 110% = TDG Water Quality Standard) (Continued)



c. Priest Rapids, Grand Coulee, and international boundary stations, 1991

Figure 2. (Concluded)

Intense regional pressure is presently being placed on the Corps to draw down the lower Snake and lower Columbia reservoirs to minimum pool elevations or below, to increase water flow velocities and thus improve juvenile fish survival. Extreme drawdowns could render generation units inoperable, requiring 100-percent spill for extended periods of time to pass inflows. A test drawdown to 30 ft below normal minimum forebay elevation is under way at Lower Granite Dam, including spill tests and TDG monitoring. This testing will require the full month of March 1992 to complete. Nitrogen supersaturation is a major concern with drawdown operational scenarios.

Fish agencies tend to show little concern for TDG until levels become extremely high. As a result, the Corps often takes the lead in showing environmental concern in this water quality arena. Ironically, adherence to a legal Water Quality Standard, which was originally based primarily on salmonid impacts in the Columbia River basin, has been compromised in practice by providing high levels of project spill, presumably to benefit these same fish. An additional impact of high spill includes adult migration delays, which have been observed on the lower Snake River (Turner, Kuskie, and Kostow 1983). Also on the lower Snake River in higher flow years, juvenile fish injury rates as recorded at the Little Goose Dam collection and transportation facilities have increased coincident with spill increases at Lower Granite, the next dam upstream (personal communication, Sarah Wik, Walla Walla District).

Project fish survival benefits may be partially to completely offset by TDG levels exceeding 120 percent for extended periods. The break-even point is an unknown, dynamic

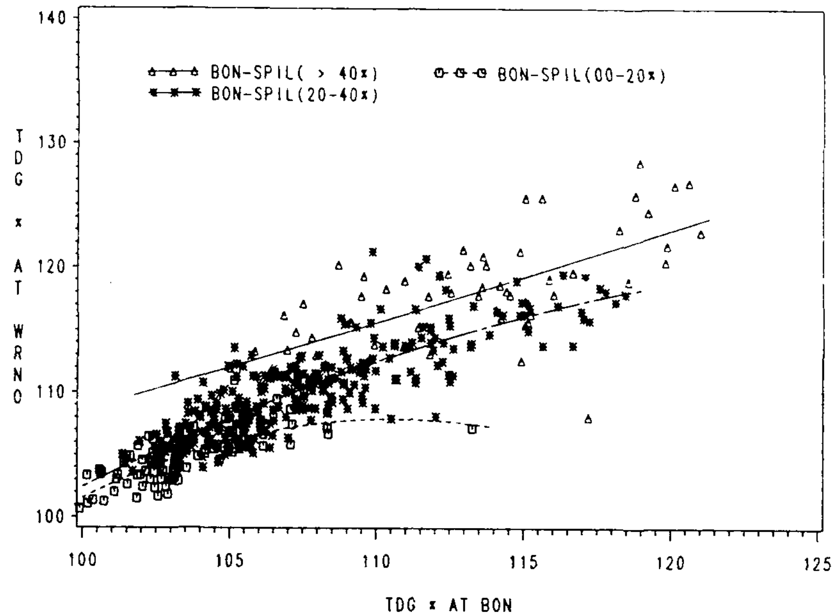


Figure 3. Dissolved gas dissipation between Bonneville and Warrendale stations, 1986-1991. Quadratic regression curves shown for <20% (n=89, r^2=0.60); 20-40% (n=312, r^2=0.72); and >40% (n=63, r^2=0.48) spill levels at Bonneville

quantity. It varies with TDG level, water temperature, fish condition, fish avoidance behavior, swimming depth, predator abundance in the tailrace, and other factors, all of which are themselves temporally and spatially dynamic. At this time, TDG is managed to prevent levels from chronically exceeding 120 percent. Immediate action is taken to limit project spill or to shift overgeneration spill to other parts of the hydroelectric system when TDG approaches 130 percent for more than several hours per day.

Currently, the Corps is in a standoff with the USEPA and the States, since the state environmental quality agencies that are responsible for enforcement of the Standards have been unwilling to press the compliance issue in spite of USEPA's concern about the regional dissolved gas situation. In 1992, the Corps intends to implement further NPPC measures calling for specific percentages of fish passage through nonturbine routes. This will include spill levels at Bonneville even higher than those now resulting from second powerhouse restrictions. Other projects may eventually be affected as well. There will be an increased need to closely manage system TDG to avoid a situation in which, by attempting to improve main stem passage survival, fish are damaged by overly ambitious project spill programs.

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Temporal and Spatial Variability of Riverine Water Quality in a Rural, Agricultural Drainage Basin

by
Steven L. Ashby[1]

Introduction

Flood control measures, such as the construction of weirs, channel clearing, and rerouting of water, have been proposed for the Steele Bayou (SB), Mississippi, drainage basin. Proposed flood control measures are anticipated to alter hydrologic processes and potentially impact water quality in the basin. These flood control measures may also provide opportunities for environmental enhancements in the basin. However, water quality information for evaluation of potential impacts on water quality is sparse. This study was conducted to describe distribution patterns of water quality and to evaluate processes relative to water quality in different locations of the study area. Data from this study can then be used in the assessment of impacts from the proposed project.

Site Description

The upper watershed of Steele Bayou is an intensively agricultural region within the historic floodplain of the Mississippi River and has undergone extensive removal of riparian vegetation. Major drainages include Main Canal (MC), which is approximately 30 km long with extensive channelization and little recent maintenance; Black Bayou (BB), which is approximately 50 km long and has had little change in channel morphometry; and the upstream reach of Steele Bayou, which is approximately 35 km long and has had extensive channelization and one-bank clearing. The downstream boundary of the study area is delineated by the only weir in the study area, which is located on the Steele Bayou drainage (Station SBS1) and results in a pool during low-flow conditions.

Water quality in the Steele Bayou basin is generally considered to be poor as a result of excessive levels of turbidity, suspended materials, and nutrients; however, historical water quality data for the upper region are sparse. Little is known about the influence of physical features, such as the type and extent of riparian vegetation, channel morphometry, and land use of the immediate drainage area, on water quality in Steele Bayou. Additionally, dynamic hydrologic conditions influence water quality and material transport in the basin (Ashby et al. 1991).

Methods

Water quality studies were conducted from March 1990 through February 1991. Studies included monthly monitoring of physicochemical constituents at 11 routine stations, seasonal intensive surveys at additional stations, and supplemental studies conducted during the summer growing season. Routine sampling stations were located along Black Bayou, Main

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Canal, and the upper reach of Steele Bayou (Figure 1). Coincident with routine sampling in June, phytoplankton photosynthesis and respiration were determined using the light/dark bottle technique (American Public Health Administration (APHA) 1980) at Stations SBS1 (open), BBS1 (canopied), and BBS3 (open). Dissolved oxygen (DO) concentrations were determined with the azide modification of the Winkler titration method (APHA 1980). Gross productivity was calculated for each station according to Vollenweider (1969). Samples for routine water quality analysis and phytoplankton identification were collected at the onset of incubation. Diel changes in temperature, DO, pH, and specific conductance were measured at selected stations in August.

Selected stations represented a weir pool, SBS1; a reach upstream from a weir pool and downstream from a recently channelized reach, SBS3; and two locations along an unchannelized reach through a canopied area, SBS5 and BBS1. Sample collection methods, analytical procedures (using standard methods--APHA 1980 and U.S. Geological Survey 1989), and complete analytical methods are reported in Ashby et al. (1991).

Results

Hydrologic conditions during the study period are depicted in the hydrograph for Steele Bayou at Station SBS1 (Figure 2). Data were grouped by high and low flow, as indicated by change in stage height. The low-flow season was defined as June through November, and the high-flow season was defined as March through May and December through February.

General water quality distribution

Highest temperatures occurred in June, July, and August, and lowest temperatures were observed in December, January, and February. Dissolved oxygen concentrations were generally above 4.0 mg/L. Specific conductance displayed highest values during summer low-flow conditions, and lowest values occurred during high-flow seasons. Highest turbidity values occurred coincident with periods of high flow and runoff. Generally, total solids concentrations were relatively constant (300 to 600 mg/L) except for increased values observed during periods of high flow. Suspended solids comprised the majority of the total solids during high-flow periods. Conversely, during periods of low flow, suspended solids decreased, and dissolved solids comprised the majority of the total solids. Higher total phosphorus concentrations generally occurred coincident with periods of elevated flows. Total soluble phosphorus concentrations were relatively constant; however, concentrations greater than 0.4 mg/L were observed at Stations SBS5, BBS3, and MCS3.5. Total organic nitrogen concentrations were generally between 1 and 4 mg/L, with values between 6 and 10 mg/L occurring in each drainage during July and October. Nitrate/nitrite nitrogen concentrations did not exceed 2.7 mg/L, and concentrations were occasionally below the detection limit (0.02 mg/L). Nitrate/nitrite values above 2.0 mg/L were observed in the Steele Bayou drainage in November and in the Black Bayou drainage in May and July. Ammonia concentrations were less than 1.0 mg/L with occasional values below the detection limit (0.01 mg/L); highest values were observed at MCS1.5, MCS3, and GBS1 in July, September, and November. Total organic carbon concentrations displayed a trend of increasing concentrations from March through December. Dissolved organic carbon comprised the majority of the total organic carbon and displayed similar concentration distribution patterns. Chlorophyll *a* concentrations displayed seasonal trends with relatively lower concentrations occurring December through May and higher concentrations occurring June through November. Chlorophyll *a* concentrations were more

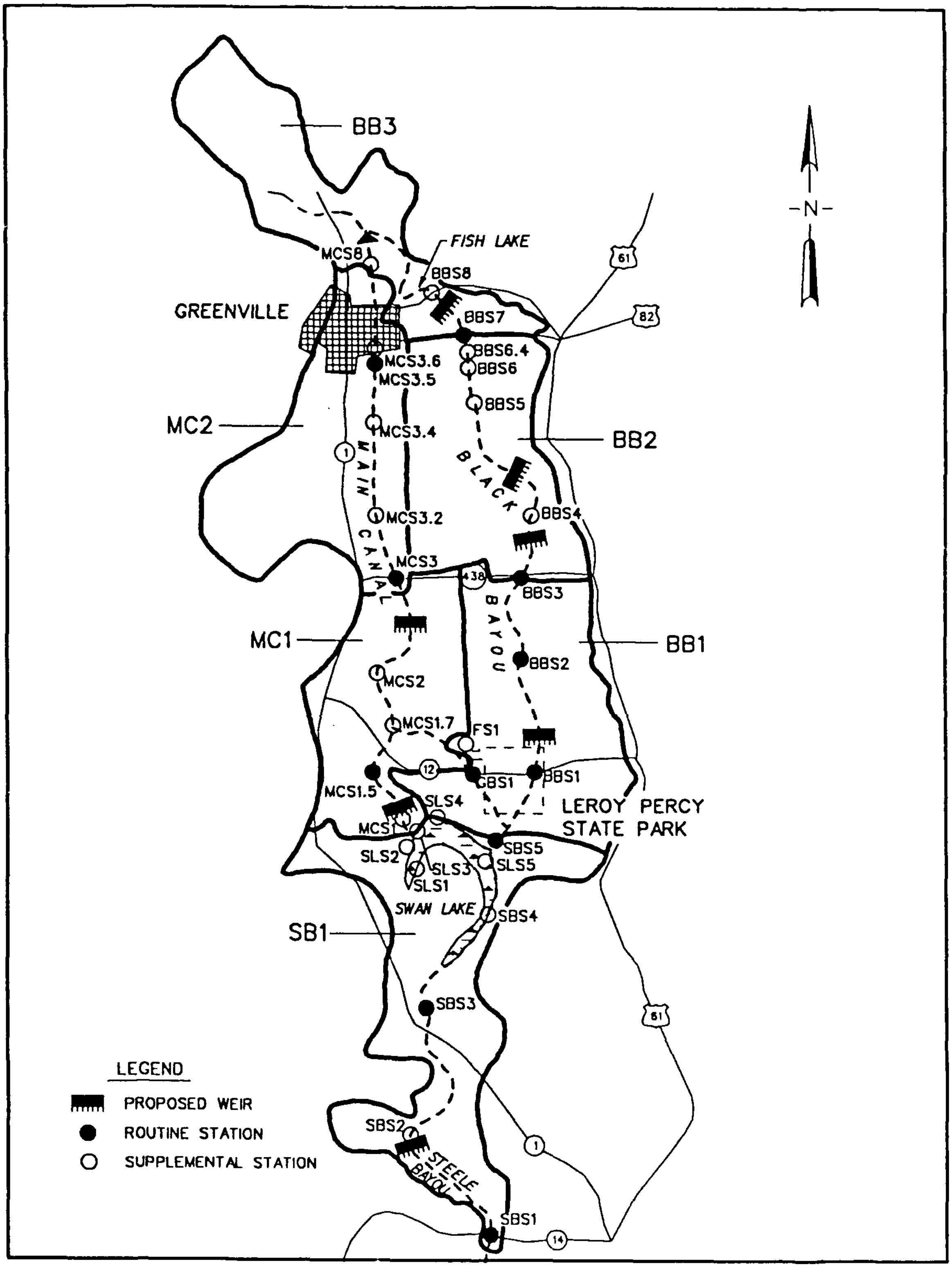

Figure 1. Major drainages and station locations in upper Steele Bayou watershed. Solid circles denote stations sampled routinely, and hollow circles denote stations sampled during supplemental studies

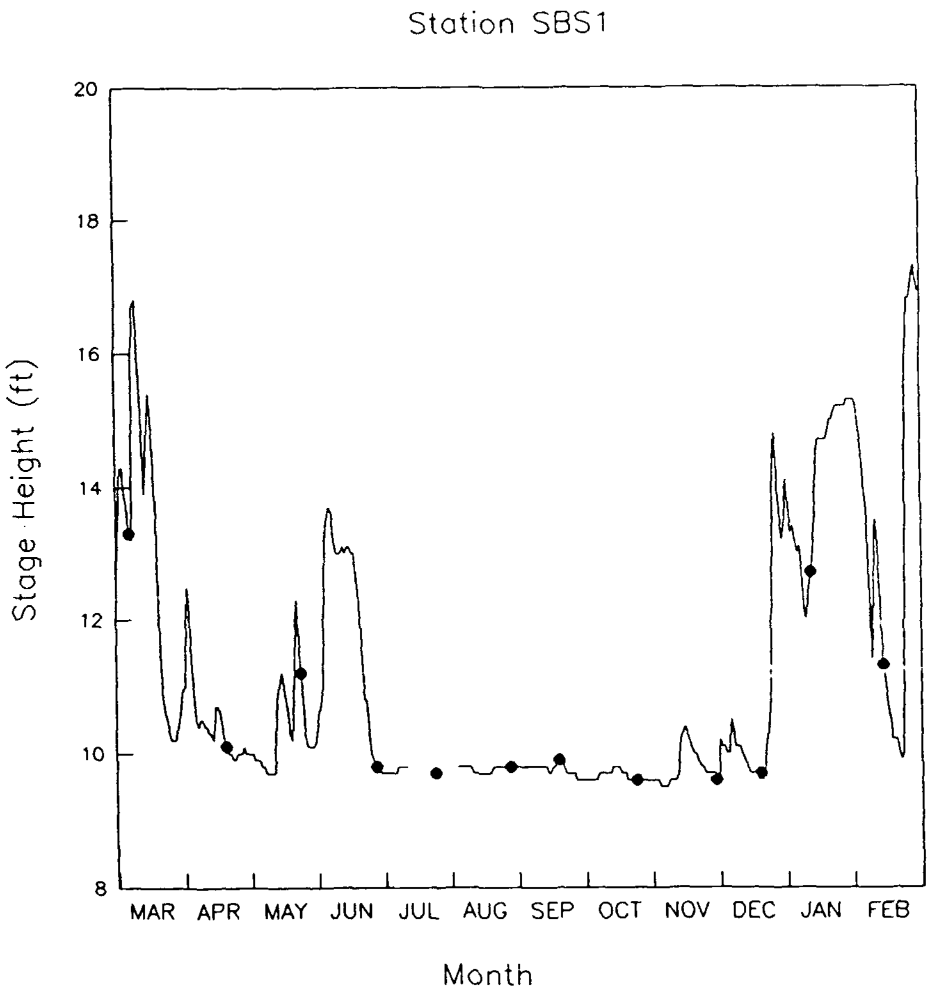

Figure 2. Stage height at downstream station during the study period. Solid circles denote routine sample collection

variable during June through November and exceeded 20 μg/L at nearly all stations at some time during the summer growing season. Complete results are reported in Ashby et al. (1991).

Spatial and temporal variability in water quality

Assessments of spatial variability in physicochemical constituents among drainages were based on annual, high-flow, and low-flow mean values. Maximum annual mean values of most constituents occurred in the Black Bayou drainage. Maximum mean concentrations of total and soluble phosphorus occurred in the Main Canal drainage, and maximum mean temperatures occurred in the Steele Bayou drainage. Annual mean concentrations of total and soluble phosphorus were significantly higher ($p < 0.05$) in the Main Canal and Black Bayou than in the Steele Bayou drainage. The annual mean concentration of total organic carbon was significantly higher ($p < 0.05$) in the Black Bayou drainage. Significant longitudinal gradients ($p < 0.05$) in annual mean concentrations at routine stations were observed only for

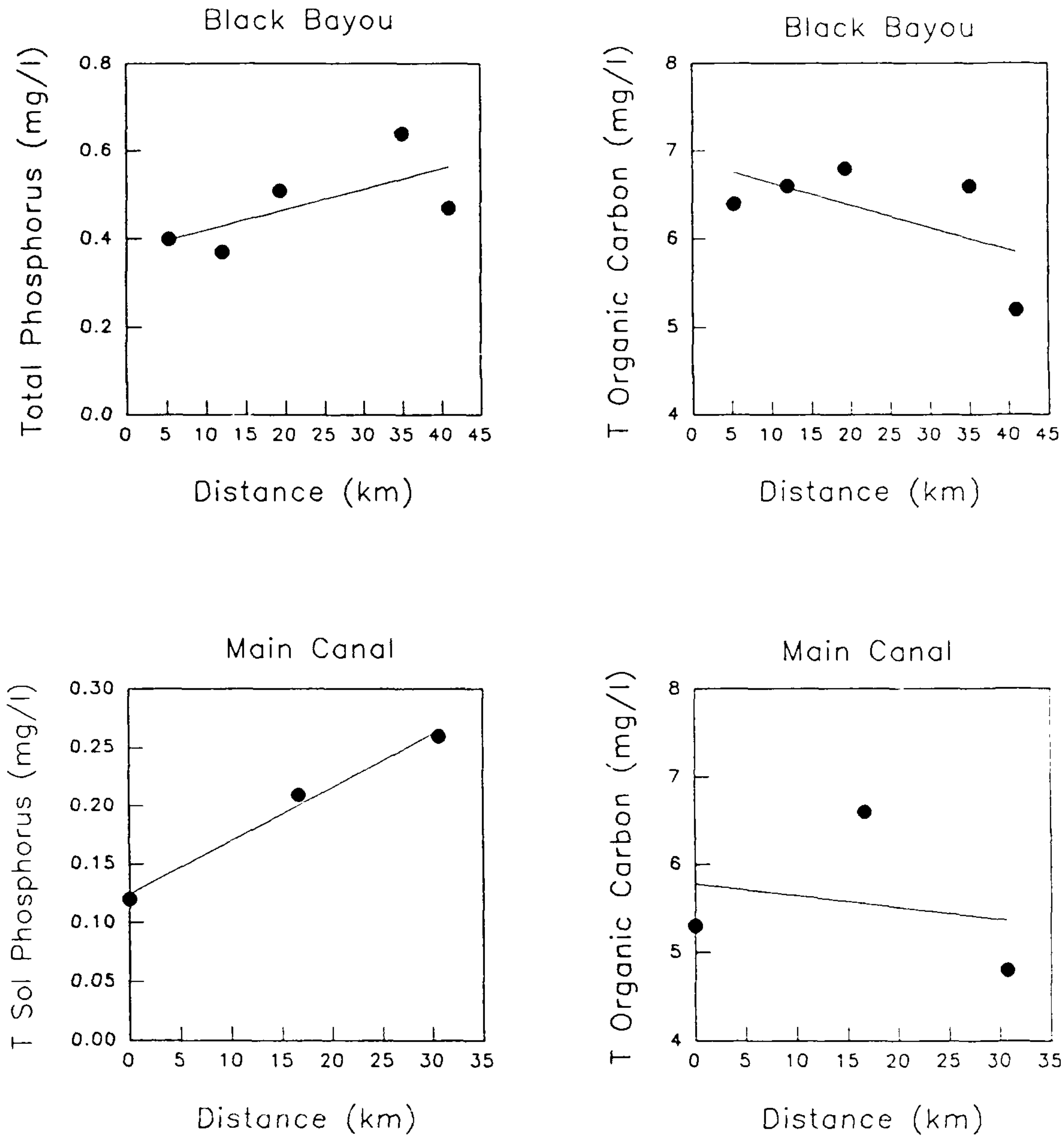

Figure 3. Annual mean concentrations with significant ($p < 0.05$) longitudinal gradients

total phosphorus and total organic carbon in Black Bayou and for total soluble phosphorus and total organic carbon in Main Canal (Figure 3). In both watersheds, phosphorus concentrations decreased in the downstream direction, and organic carbon increased in the downstream direction.

Mean concentrations for the high- and low-flow seasons are presented in Figure 4. Mean concentrations within each drainage were generally highly variable (coefficient of variation greater than 50 percent), with less than 30 percent variability being observed only for organic carbon, and were often significantly different ($p < 0.05$). Mean values outside the limits for the 25th and 75th percentile may be attributed to occasional high values associated with high-flow events (i.e., chlorophyll *a*).

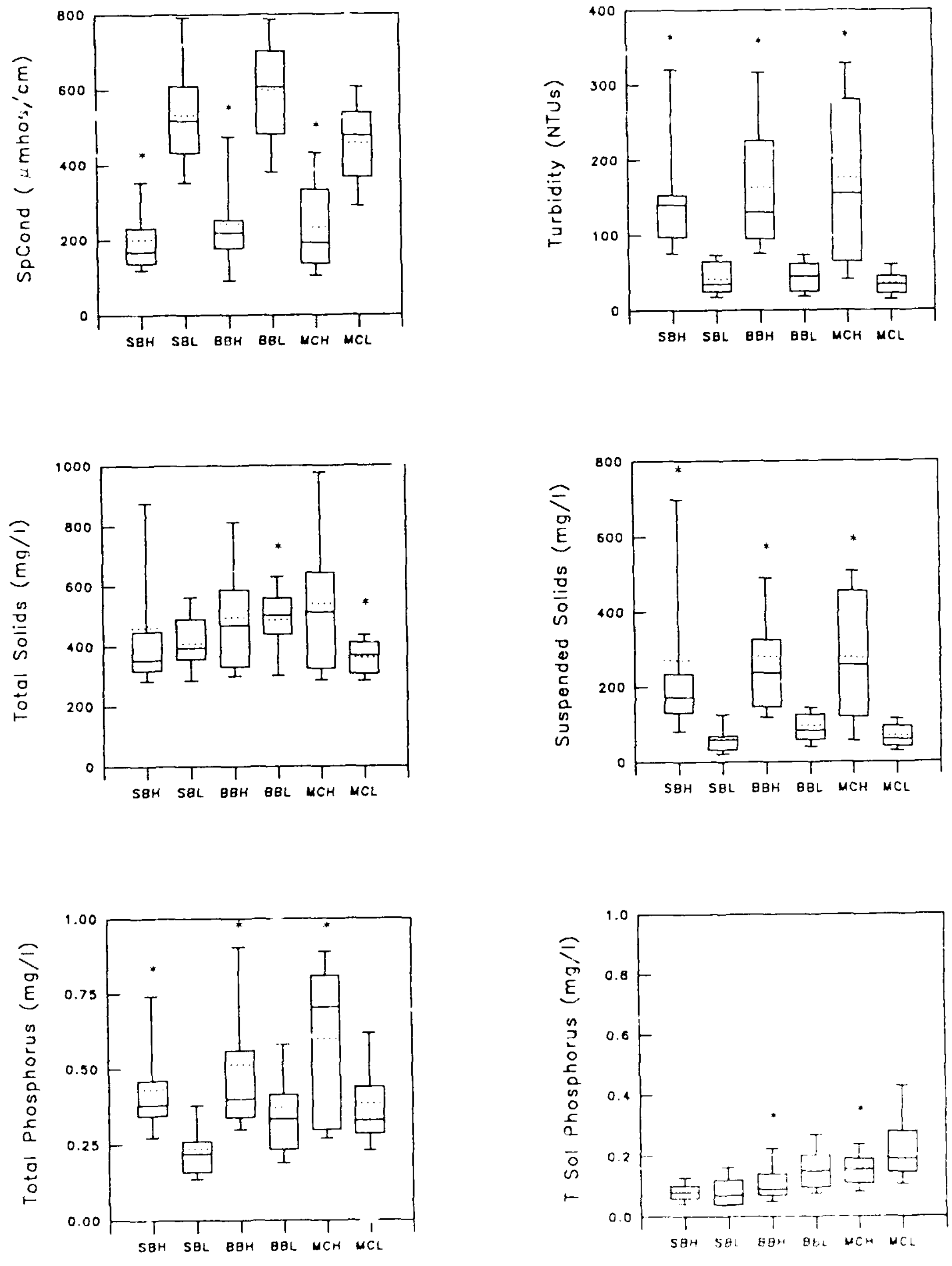

Figure 4. Concentration distribution by hydrologic season within each drainage. (H and L denote high- and low-flow seasons, respectively. Dotted line is mean value, solid line is median value, boxes represent 25th and 75th percentiles, outer error bars represent 10th and 90th percentiles, and asterisk denotes that mean values in the watershed were significantly different ($p < 0.05$) between seasons (Continued)

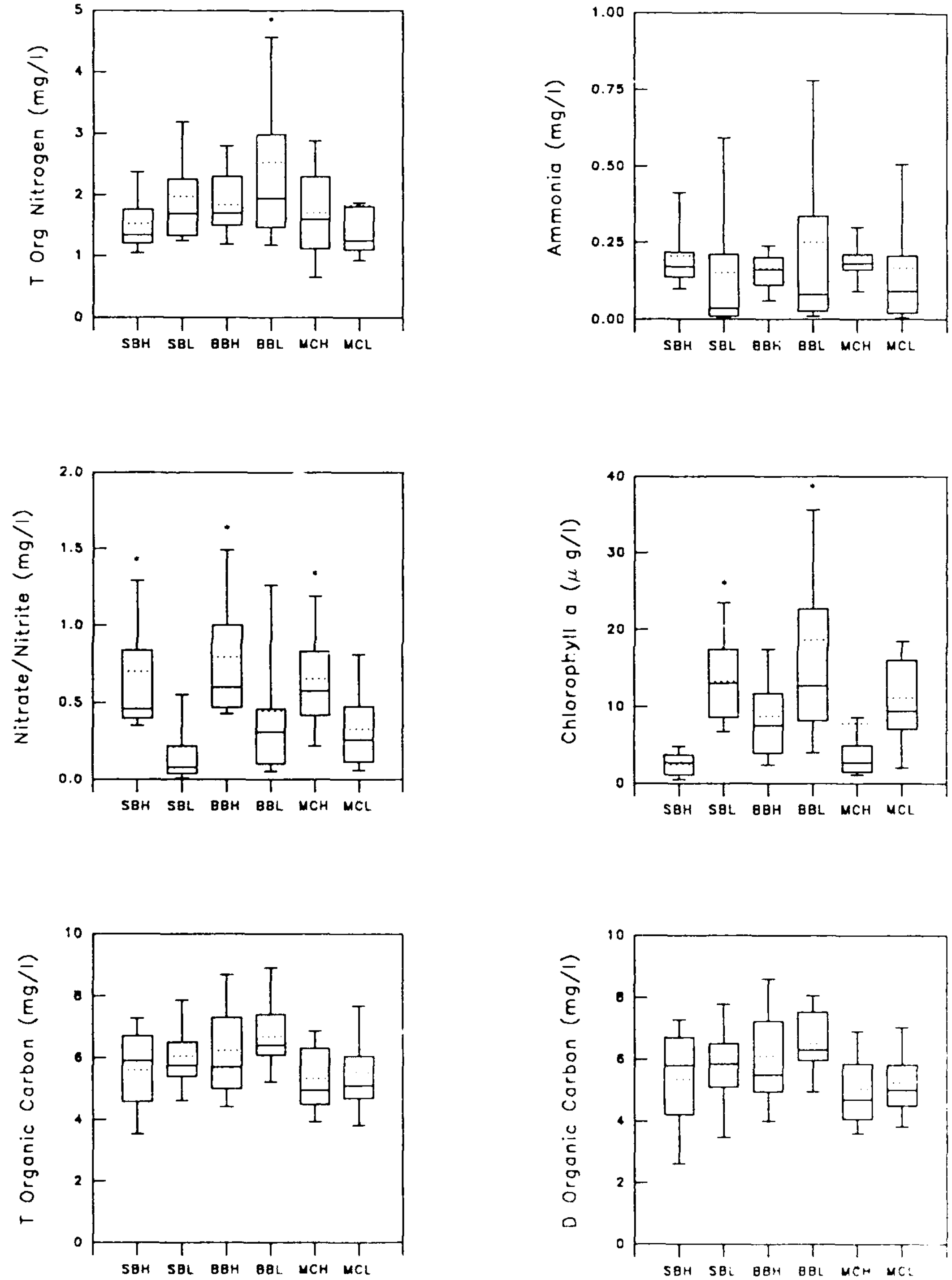

Figure 4. (Concluded)

Assessment of canopy-covered and open reaches

Differences between canopied (Station BBS1) and open reaches (Stations SBS1 and BBS2) were apparent in evaluations of the phytoplankton community. Rates of net and gross photosynthesis were similar at stations in open reaches and considerably higher than rates at the station in the canopied reach. Dissimilarly, respiration rates decreased in a downstream direction. Daily gross productivity was higher in the open reaches, Stations SBS1 and BBS2, than in the canopied reach, Station BBS1. Conversely, chlorophyll *a* values at Station BBS1 were approximately twice those at Stations SBS1 and BBS2. Blue-green algal species (e.g., *Cyclotella*, *Oscillatoria*, and *Microcystis*) were dominant at the time of the study. Specific conductance, total solids, suspended solids, and total phosphorus were higher at BBS1 than at SBS1 and BBS2. Total soluble phosphorus was relatively lower at SBS1. Total organic carbon and total nitrogen concentrations decreased from BBS2 to SBS1.

Assessment of diel variation of in situ constituents

Diel variations in temperature and DO were apparent at all stations in the downstream region of the study area (Figure 5). Temperature maxima occurred in late afternoons and were between 28 and 30 °C, with highest values occurring at SBS1. Temperature decreases by approximately 2 °C were observed in early mornings. Dissolved oxygen maxima and minima concentrations occurred in late afternoons and early mornings, respectively. While maximum concentrations were between 7 and 9 mg/L at SBS3, SBS5, and BBS1, highest concentrations occurred at SBS1 (Steele Bayou at Rolling Fork) (> 10 mg/L). Minimum values remained above 3.5 mg/L at all stations.

Discussion

Spatial and temporal patterns in constituent concentrations may be attributed to influences of hydrologic events, channel morphometry, hydrology, and local land uses. Influences of hydrologic events were most obvious, with elevated concentrations of suspended material coincident with precipitation and runoff events. Conditions of high turbidity, total phosphorus, and suspended solids were most prevalent during the wet season (late fall to early spring). Consequently, transport and loading of surface pollutants was greater during periods of runoff. Conversely, specific conductivity values were highest during summer low-flow periods, indicating dissolved constituents from biological and hydrologic sources (groundwater and agricultural dewatering practices) markedly contributed to seasonal water quality.

During low-flow periods, highly variable distributions of selected constituent concentrations suggest that local conditions markedly influence water quality. Local conditions affecting water quality include increased hydraulic retention and accumulation of settling material, as a result of decreased velocities during low flow, extent of riparian vegetation, and land use of the surrounding watershed. Significantly higher concentrations of solids in the downstream region of Black Bayou suggest accumulation of material from local and upstream subwatersheds. Small-grained sediment remaining in suspension and dense algal populations probably contributed to elevated suspended solids concentrations. Higher concentrations of chlorophyll *a* in the downstream region of the watershed during low flow suggest increased biological processing of nutrients. In situ measurements of temperature and DO indicated vertical gradients at the weir pool (Station SBS1) during the study conducted in June. Increased

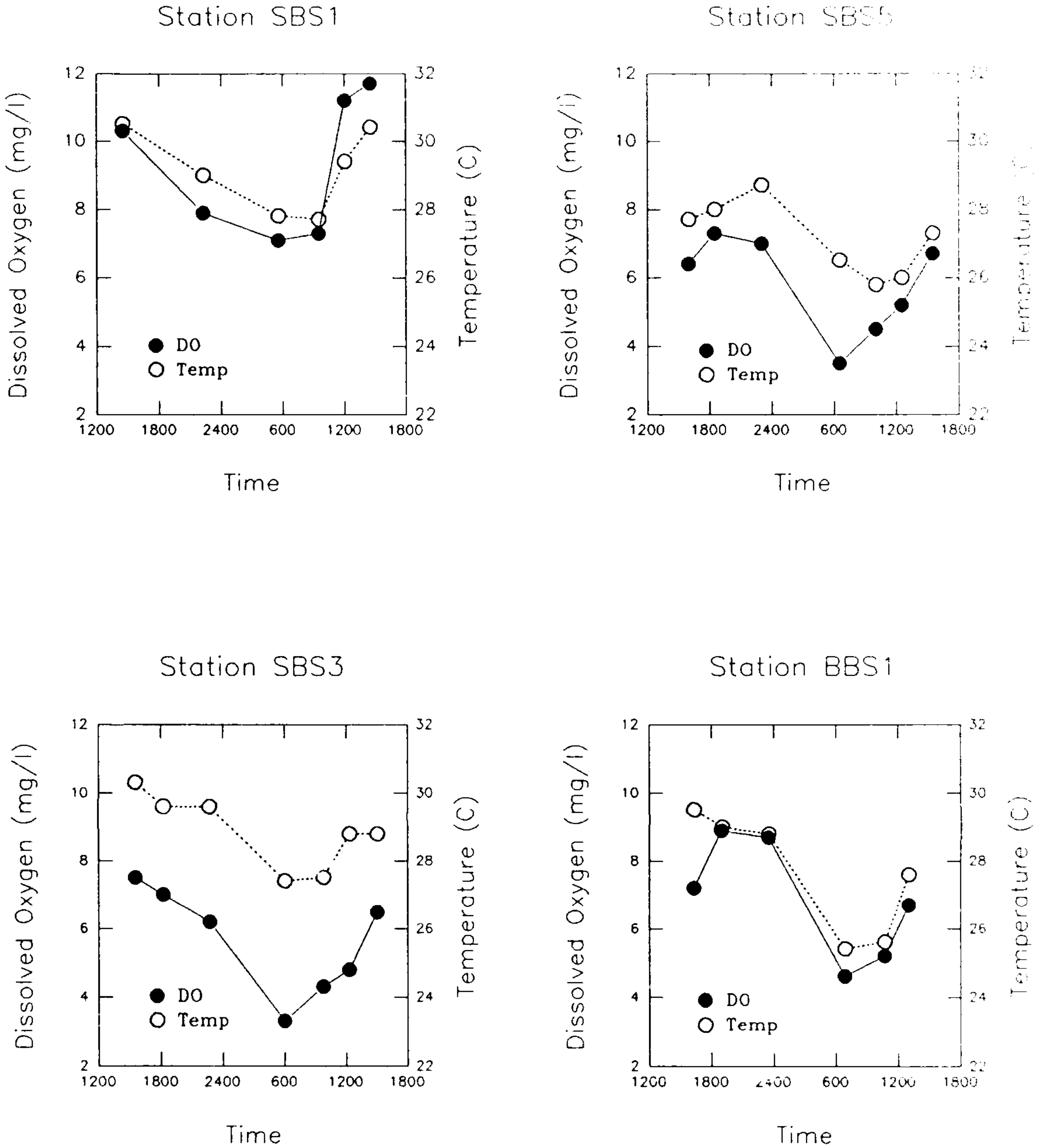

Figure 5. Diel variations in temperature and DO values during low-flow conditions at stations with different stream morphometry and extent of riparian vegetation

thermal structure would be expected with increased hydraulic retention time in a weir pool resulting in a heat gain, particularly in turbid waters.

Selected physicochemical and biological constituents displayed pronounced distribution patterns. Diel variations of DO concentrations may be attributed to a highly productive phytoplankton community in the downstream region of the study area. Chlorophyll *a* concentrations greater than 20 μg/L during the summer growing season are further evidence of a highly productive system, particularly in the downstream regions of Main Canal, Black Bayou, and Steele Bayou. Excessive chlorophyll *a* concentrations may be attributed to excessive

phosphorus concentrations. Excessive populations of "nuisance" blue-green algae, which can obtain the necessary nitrogen from the atmosphere (Sakamoto 1966), may be attributed to excess phosphorus and low nitrogen concentrations. An organic-rich system is indicated by relatively high dissolved organic carbon concentrations. Pronounced material transport is suggested by relatively high values of total solids and total suspended solids.

Local influences of riparian vegetation were apparent in evaluation of the phytoplankton community. The presence of a vegetation canopy decreases light availability (Hill and Harvey 1990) and impacts the quality of local runoff (Vitousek and Reiners 1975). In the canopied reach near Leroy Percy State Park, phytoplankton productivity was lower, and chlorophyll *a* concentrations were higher when compared to open reaches.

Conclusions

Sediment and nutrient concentrations are relatively high in the major drainages of the upper Steele Bayou drainage. Elevated sediment and nutrient concentrations may be attributed to anthropogenic influences associated with intensive agricultural practices. As a result of the hydrologic conditions and seasonal agricultural practices, concentrations and longitudinal distributions are highly variable. With the exception of total and soluble phosphorus and total organic carbon, spatial distribution among streams was not pronounced annually. Temporal variation was most pronounced in association with hydrologic conditions (dry and wet seasons) within drainages. The different responses in temporal variation of physicochemical and biological constituents within each drainage suggest that spatial variability among drainages may exist on a seasonal basis. Diel measurements and estimates of primary productivity, although limited to one sampling event each, further indicate that spatial variations exist among these drainages. Spatial variability may be attributed to varied channel morphometry, extent of riparian vegetation, and the hydrology of low-flow conditions. Potential impacts to water quality may not be discernible because of the highly variable temporal and spatial distribution patterns of selected water quality constituents.

Acknowledgments

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Water Quality Data Management

by
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Need for Data Management

The need to manage water quality data should have been evident when the plans for collecting and utilizing the data were first made. Unfortunately, the ability to collect samples for analysis and field measurements probably exceeded that of data management in the early days of Corps interest in water quality. Manual records in filing cabinets may have been the typical storage method long after computers were being used in engineering calculations. Although manual records with signatures are still needed for some legal situations, the current volume of our water quality data dictates an automated system for storage and retrieval. The need to display data graphically and analyze data mathematically makes computer files the ideal storage method.

Evolution of Storage Methods

The availability of computer hardware and software has greatly influenced storage methodology. The earliest automated storage of water quality data utilized paper tape or cards as a storage media. Cards even had the advantage of "random access"; that is, the deck could be shuffled or sorted! Later systems used magnetic tape that could be accessed sequentially. The later use of magnetic disk drives made random access viable. This random access capability led to the development of the relational database concept in which a group of files containing dissimilar data could be "related" to each other by certain key elements. This concept greatly reduced storage space and processing requirements since repetitious information could be stored in auxiliary files and accessed only when needed.

History of Water Quality Databases in the Ohio River Division (ORD)

ORD has been active in the automated management of water quality data since the late 1960s. A variety of systems and hardware platforms ranging from General Electric computers in the early years to mini- and micro-computers have been used. Prior to 1973, water quality data were stored in various forms in each District on cards or magnetic tape.

ORD LABMASTER

In 1973 the development of the first Division-wide system for storing water quality data from the ORD Water Quality Laboratory was begun. This first system (LABMASTER) used a sequential file structure on magnetic tape for results and abbreviated identification information along with a few disk-resident tables of test names and financial information. This system was ported from the original General Electric system to a time-sharing system using Univac equipment (CSC INFONET), to Honeywell systems in both ORD and the South

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Atlantic Division, and still later to two generations of Harris mini-computers. Eventually, this system became a complex system of related data files that served the laboratory information management system needs of the ORD Laboratory until 1991 when it was replaced by a Perkin Elmer LIMS.

ORD AURAS

While LABMASTER met the needs of the ORD Laboratory, it was not a good storage and retrieval system for District needs. The ORD Automated Upward Reporting and Analysis System (AURAS) was developed on CSC INFONET in the late 1970s. The bulk of the data were stored in sequential files on disk. Data were stored in order of location, time, depth, and parameter with the rest of each record containing other information such as laboratory number, test result, test cost, etc. EPA STORET's system of defining test parameter numbers was used to facilitate data exchange. A separate sequential file containing valid parameter numbers and information about them was maintained. An index-sequential file containing location descriptions compatible with STORET could be accessed randomly or sequentially. A file containing a table of office names and single-character codes was used to reduce storage volume. CSC INFONET provided a means for storing data for each project in a single file and conveniently combining the project files for updates of the entire database. This made retrievals and reporting much less expensive, since only data for the desired project had to be read. AURAS was ported to Harris mini-computers in the early 1980s and is now being ported to a personal computer (PC) with an SCO UNIX operating system. AURAS has interfaces to other databases such as STORET, HEC DSS, and various DOS PC applications.

HEC DSS

The Corps Hydrologic Engineering Center's (HEC) Data Storage System (DSS) may be the best and least expensive method to store time-series data. It has excellent graphics capability, and its library of retrieval and storage subroutines makes it ideal for using data in modeling applications. For water quality data collected by robot monitors or other sampling methods in which many observations of the same parameter are made with time as the primary variant, DSS is the outstanding choice of storage methods. Unfortunately, it may not be very efficient when depth is the primary variant and/or when samples are collected at many locations a few times each year. Additionally, it lacks a good method to store location-descriptive information and detailed parameter definitions. In ORD it is used for storage of robot monitor data, and in its paired data form, it is used to provide graphic displays of profiles after data are ported from AURAS.

EPA STORET

It is policy in ORD that all appropriate water quality data in ORD be stored on the EPA STORET system for use by non-Corps users as well as ORD users. After Districts have screened data for errors, previously untransmitted data are sent to STORET by magnetic tape a few times each year. The major problem with storing data on STORET is that each location must be completely and properly defined before data for that location can be stored. Routine storage operations with AURAS are much easier because of its tolerance for partial or nonexistent location description before storage of results. Even with storage difficulties, STORET provides comprehensive access to Corps data along with data stored by other agencies.

Commercial databases

Circa 1980, ORD considered using a commercial hierarchial database called System 2000 on CSC INFONET. It soon became apparent that the cost of disk storage for an online database was too great, and implementation was never begun. Today, the cost of disk space on Corps-owned computers has made online storage much more economical, particularly on PCs. Commercial packages such as dBase running on PCs are far more user-friendly than their mainframe counterparts. ORD users have downloaded subsets of data from AURAS and stored them in PC databases and spreadsheets. Some packages provide multiuser access on a local area network with record locking for multiple concurrent updating. Additionally, some packages, such as Paradox, provide excellent full-screen QBE (Query by Example) capability, and at additional cost, an interface to a SQL server (an ANSI industry standard query language for relational databases).

This methodology has given ORD personnel convenient access to various mathematical analysis and graphical packages. This is an excellent way to manipulate small to medium datasets, but it may not be very efficient when working with databases containing millions of observations and associated data unless a sophisticated and probably expensive database package is employed. Additionally, the extreme ease of database modification makes the loss of data a distinct possibility. Unless strict controls on database modification are available and utilized, a database on a DOS PC or network server should not be the primary storage system.

Commercial environmental databases

ORD has participated with HEC in testing a commercial environmental database called EQUIS, which was developed by Egret Technologies. While this comprehensive package may be overkill for routine water quality storage requirements, its capability to manage data for hazardous and toxic wastes (HTW) along with water quality data is a significant advantage when our water quality staff involvement with HTW activities is considered. It uses ORACLE, which can run on a variety of hardware platforms and under a variety of operating systems. Since ORACLE supports SQL and various programming language interfaces for storage and retrieval operations, it may be possible to use it in models along with DSS for time-series data. Its major drawback is its high cost both in software acquisition and the manpower required to use it properly.

Need for a Standard Relational Database

Those involved in the development of the next generation of Corps water control software are aware of needs for a relational database that cannot be met with DSS. Unfortunately, there does not seem to be a high priority placed on selecting or developing a system to meet these needs. We in the water quality community could choose an industry standard database system, such as ORACLE or one of its competitors, and it is **possible** that the rest of the water control users would make use of that system for general water control needs. The acquisition of a complex database system from a commercial vendor is expensive in terms of dollars spent and in the effort to learn and manage the system. We may be better off using a low-cost hybrid system until there is better direction for a standard relational database system for water control in general.

Recommendations

Although existing systems such as AURAS might be expanded to handle data from 30 to 40 offices, I believe that it is time that the Corps develops or acquires a relational database system with SQL or full-screen QBE capabilities. The system should provide for automated updating (additions, modifications, and deletions) in batch in addition to full-screen editing and data entry to facilitate corrections and additions in a user-friendly environment. Programming language interfaces should be available to support modeling uses. Furthermore, this database system should be used in conjunction with DSS for all water management applications, including water quality. This implies that the system should be available on the same platforms used for other water management activities. The system should make ad hoc queries very easy and provide a facility for storing frequently used queries and reports. The system should be appropriately economical to acquire and use. These last two requirements may lead to a hybrid system, with an inexpensive DOS database package providing the user interface for queries, and a simple sequential file storage system on a UNIX platform providing the secure database updating capabilities. The decision for acquisition from a commercial source or internal development as with DSS is not an easy one, considering the time and manpower required to develop and maintain a system.

Water Quality Data Elements

Whatever database is used for water quality, it should include tables (files) for the following major functions:

a. Location--complete description of location including latitude, longitude, river mile, state, county, etc.

b. Sample--complete description of sample, including location, depth(s), times of acquisition, preservatives, times of analysis, etc.

c. Results--along with descriptive information such as parameter, location, sample identification, etc.

d. Parameter--complete description of the testing process along with units, detection limits, etc.

e. Various tables with names associated with state, county, basin codes, etc.

Standard data elements for water quality

If a single relational database system is to be used for storing water quality data (robot monitor data excluded), then we as an organization must agree on the required data elements along with their sizes and relationships. Decisions about the structure of the database are not easy. For instance, a structure with several results for the same sample displayed side by side has advantages when working with a spreadsheet, but it may not be the most efficient form for storage when hundreds of parameters are present in the database.

Even if we are unable to agree on a single relational database, there are advantages in agreeing on database elements and relationships. We must decide if data for HTW and other

specialized sampling programs should be included and what additional data elements are then required. We must also tackle the problem of quality control (QC) and assurance data, and how they fit into the database.

We must be careful in choosing the size of data elements that will appear frequently in the database. "There is no free lunch"; that is, if we make some data elements unnecessarily wide or include unnecessary ones, the total size of the database will make storage and processing difficult and/or expensive. The suggested elements and sizes in the following paragraphs should be reviewed and revised to meet the needs of all Corps water quality elements.

Suggested location data elements

The location information must be sufficient to provide for automated exchange with other agencies such as STORET. Since the location identifier will appear on all samples, it should be small to reduce storage requirements, but still large enough to permit an easily recognizable structure to facilitate retrievals of related locations. The primary location identifier should contain a maximum of 15 bytes, starting with a three-character office symbol such as "ORH" followed by a 3- to 5-byte project symbol. To be compatible with STORET, the following elements should be available for each location (numbers in parentheses show present ORD AURAS sizes):

a. Primary location	-	15 bytes	(9)
b. Alias 1	-	12 bytes	(9)
c. Alias 2	-	12 bytes	(9)
d. Alias 3	-	10 bytes	(9)
e. FIPS state code	-	2 bytes	(2)
f. FIPS county code	-	3 bytes	(3)
g. Latitude (DDMMSST)	-	7 bytes	(7)
h. Longitude (DDDMMSST)	-	8 bytes	(8)
i. Precision of lat./long. code	-	1 byte	(1)
j. Depth units (feet or meters)	-	1 byte	(1)
k. Maximum depth	-	3 bytes	(3)
l. Location basin codes	-	6 bytes	(6)
m. Brief description	-	48 bytes	(48)
n. Major basin name	-	24 bytes	(24)
o. Minor basin name	-	40 bytes	(40)
p. Variable length description	-	1,000 bytes	(1,000)
q. Present on STORET flag	-	1 byte	(2)
r. Elevation	-	5 bytes	

Suggested sample data elements

The sample entries should provide sufficient information to identify where and when the sample was collected. The elements below marked with an asterisk may be more important for HTW than for normal water quality sampling activities. Sizes of numeric fields may be reduced if the database provides a more efficient method for storing numbers. The following elements are suggested:

a. Sample number	-	8 bytes	(5)
b. Depth units (feet or meters)	-	1 byte	(feet only)

c.	Depth (decimal permitted)	- 8 bytes	(3)
d.	Sampling time (MMDDYYHHMMSS) (dates beyond 1999 permitted)	- 12 bytes	(10)
e.	Composite code	- 2 bytes	
f.	Composite depth (0 def.)	- 8 bytes	
g.	Source code	- 2 bytes	
h.	Collection method	- 2 bytes	
i.	Field batch number	- 8 bytes	
j.	Alternate sample number	- 8 bytes	
k.	Physical state code*	- 1 byte	
l.	Media code*	- 2 bytes	

Laboratory sample data elements

Since it is possible for samples to be split and sent for analysis to multiple laboratories or measured in the field, it may be necessary to have a separate table for laboratory samples, in order to completely define the testing process. In the following list, those elements marked with an asterisk seem more important in HTW efforts:

a.	Laboratory identifier	- 8 bytes	
b.	Lab sample number	- 8 bytes	(5)
c.	Lab batch number	- 8 bytes	
d.	Date received (MMDDYYHHMM)	- 10 bytes	
e.	Cleaning method*	- 4 bytes	
f.	Preparation method*	- 10 bytes	
g.	Testing method*	- 10 bytes	
h.	Extraction time* (MMDDYYHHMM)	- 10 bytes	
i.	Funding code*	- 32 bytes	

Test results data elements

In the following list of test results data elements, the ones marked with plus signs could easily be eliminated to save space, provided that there is a separate facility to manage the costs associated with testing:

a.	Field sample number	- 8 bytes	(5)
b.	Laboratory identifier	- 8 bytes	
c.	Lab sample number	- 8 bytes	(5)
d.	STORET parameter number	- 5 bytes	(5)
e.	Value	- 13 bytes	(6)
f.	Detection limit	- 13 bytes	(5)
g.	Significant digits	- 1 byte	
h.	Analysis date (MMDDYYHHMMSS)	- 12 bytes	
i.	Data confidence flag	- 1 byte	
j.	Archived flag	- 1 byte	
k.	Result flag	- 1 byte	(1)
l.	QC passed flag	- 1 byte	
m.	Confirmed flag	- 1 byte	
n.	Number of retries	- 1 byte	(1)
o.	Test cost+	- 8 bytes	(5)

p. Billed flag+	-	1 byte	(1)
q. Entry date (MMDDYY)	-	6 bytes	(6)
r. Present on STORET	-	1 byte	(1)

Test parameter numbers

Parameter numbers from STORET with supplements for new/specialized tests not yet defined by STORET should be used to define tests. The following table structure is suggested:

a. Parameter	-	5 bytes (5)
b. Units	-	8 bytes (6)
c. Short description	-	16 bytes (18)
d. Long description	-	32 bytes

Office symbols table

A table providing the names for the office symbols used in location names should contain the following:

a. Office symbol	-	3 bytes	(1)
b. Long name	-	32 bytes	(10)

Project code table

Assuming that we agree to a 5-byte project code in the location name, the following is suggested:

a. Project code	-	5 bytes
b. Long name	-	32 bytes
c. Reference latitude (DDMMSST)	-	7 bytes
d. Reference longitude (DDDMMSST)	-	8 bytes
e. Variable length commentary	-	1,000 bytes

FIPS state/county table

A table with the names of states and counties with the following elements is suggested:

a. State code	-	2 bytes
b. County code	-	3 bytes
c. State name	-	15 bytes
d. County name	-	20 bytes

Miscellaneous tables

Various tables will be needed to define such elements as preparation methods, testing methods, laboratories, confidence levels, composites, sources, physical state, media, cleaning methods, funding, data confidence levels, archive codes, result flags, confirmation flags, and basin codes. In general, these tables should have a key item equal in length to that used in other tables, along with an appropriately sized description for each entry.

Consensus

There are many options for storing water quality data in the Corps, including the enhancement of currently used methods. We can continue to go our separate ways, or we can try to develop a common method that can meet the needs of most offices. I suggest that the Corps make the corporate decision to standardize storage methodology, choose the most appropriate method, and begin implementation. This decision should include field input for database elements and organization to make the system most useful to those who will be using it most.

Assessment of Water Quality Patterns Using Landsat Images

by
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Robert K. Feeney,[2] and Diane I. Findley[3]

Introduction

Most reservoirs are large and morphologically complex. These features complicate water quality data collection and interpretation because cost and logistical considerations frequently limit sampling effort to a relatively few stations sampled monthly or seasonally. Such sampling designs reduce data value and overlook the importance of spatial and temporal heterogeneities in water quality (Kennedy, Thornton, and Ford 1985; Gaugush 1987).

Remote sensing techniques provide a means to describe and evaluate spatial patterns in water quality (e.g., Lira et al., in press). Such information is of particular importance for managers of reservoirs receiving nutrient loads from multiple tributaries or reservoirs potentially impacted by local land use activities. Evaluation of spatial patterns in water quality responses to external influences provides a means for developing and implementing water quality management plans. Reported here are results of efforts to describe certain water quality patterns in a large southeastern reservoir using Landsat digital image data.

Site Description

West Point Dam, located on the Chattahoochee River 120 km downstream from the city of Atlanta near West Point, GA, was completed in 1974. Authorized purposes include hydroelectric power generation, flood control, fish and wildlife development, and streamflow regulation for downstream navigation. West Point Lake is a long (53-km), narrow, dendritic impoundment (shoreline ratio of 23) with two large secondary embayments. Mean and maximum depths are 7.1 and 31 m, respectively. The primary source of water and material loadings to the lake is the Chattahoochee River; other important sources include Yellowjacket Creek, New River, Whitewater Creek, and Wehadkee Creek. Local land uses include forest, old pasture, and urban/residential and industrial development.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

[2] Murray State University, Murray, KY.

[3] US Army Engineer District, Mobile; Mobile, AL.

Methods

Data collection

Sixty water quality sampling stations were located uniformly throughout the lake. Station locations, which coincided with the locations of navigation buoys, were geographically referenced using global positioning techniques (Trimble Navigation Systems, California). When necessary, inconsistencies in geographic descriptions between sampling trips were rectified using maps and field observations of reference landmarks.

Landsat-Thematic Mapper (TM) multispectral digital images were obtained commercially (EOSAT Corporation, Lanham, MD). Landsat-TM provides seven bands of data: blue-green, green, red, near-infrared, two middle-infrared, and thermal infrared. A total of 255 digital values are possible for each 900-m^2 pixel. Image acquisition occurs at 16-day intervals.

Water quality data were collected coincident with six image acquisition dates during the period April-October 1991. Dissolved oxygen concentration and temperature were measured in situ immediately below the surface at all stations. In situ profiles were obtained at selected stations at intervals necessary to describe depth-related differences. Water clarity was quantified using a standard 20-cm Secchi disc. Surface water samples for pigment and turbidity analyses were obtained at each station; water samples for nutrient analyses were collected at selected stations. Water samples were held on ice in the dark prior to analyses.

Total phosphorus and nitrogen concentrations were determined colorimetrically following acid-digestion of unfiltered water samples (American Public Health Association 1980). Algal pigment (chlorophyll *a*, *b*, and *c*) concentrations were estimated using the trichromatic equation following extraction from glass-fiber filters using dimethyl-formamide (Speziale et al. 1984, Hains 1985). Turbidity was determined nephelometrically. Nonalgal turbidity was calculated from Secchi disc and chlorophyll *a* concentration (Walker 1982).

Data analyses

Landsat-TM digital imagery data were geographically referenced in the Universal Transverse Mercator projection using standard procedures (Marocchi 1990). Images were filtered to remove scan-line banding and stripping prior to geographic referencing (Crippen 1989, Jones and Naugle 1990). Models describing relations between image data and the observed turbidity and chlorophyll *a* concentration were evaluated using multiple linear regression analyses (SAS Institute 1988). Average band values for a 9-pixel area (8,100 m^2) centered on each sample location were used for all regression analyses.

Water quality associations among stations were identified using cluster analysis (SAS Institute 1988) following data normalization to remove the influence of differences in measurement units (Gaugush 1982). Variables included in the analysis were limited to water temperature, turbidity, and total pigment (the sum of chlorophyll *a*, *b*, and *c*). These variables correspond to those included in image analyses. Differences in water quality characteristics of resulting clusters were evaluated using Duncan's multiple range test.

Results and Discussion

Two Landsat-TM images, acquired on June 8 and September 28, 1991, were used for analyses. These dates correspond to the early and late growing season periods, respectively. Cloud cover, excessive haze, or sensor failure precluded use of images acquired on other dates.

Multiple linear regression models, based on band values and field data, account for a large percentage of the variability in turbidity on both dates (Table 1). Similar models describing chlorophyll concentration accounted for a lower percentage of the observed variability. R-squared values were 0.47 and 0.77 for the June and September models, respectively.

Table 1
Coefficients for Multiple Regression Models of Turbidity and Chlorophyll *a*

Variable	R^2	n	Model Coefficient[1] Int	B1	B2	B3	B4
			June 8, 1991				
Turbidity	0.95	47	-26.16	--	0.858	0.561	-0.698
Chlorophyll *a*	0.47	42	84.42	-1.073	--	--	1.149
			September 18, 1991				
Turbidity	0.99	43	-14.99	--	--	1.298	--
Chlorophyll *a*	0.77	39	30.47	--	--	-0.865	--

[1] Int = intercept; B1-B4 = Landsat-TM band 1-band 4.

Extrapolation of field data using these models and the digital data for each image indicates marked spatial patterns in the distribution of turbidity and chlorophyll *a*. During June, chlorophyll *a* concentrations were highest at the deeper, open-water areas in the lower portion of the pool and in the middle reaches of Wehadkee and Yellowjacket Creek embayments. Lowest chlorophyll *a* concentrations occurred in the riverine portion of the pool and near inflows from secondary tributaries. These latter areas corresponded to areas of high turbidity, indicating potential light limitation of algal growth.

Different patterns in the spatial distribution of turbidity and chlorophyll *a* were identified in September. Chlorophyll *a* concentrations were highest in coves and small embayments in the upper reaches of the lake, immediately downstream from the plunge point (as evidenced by sharply declining turbidity), and in headwater areas of Yellowjacket, Wehadkee, and Maple Creek embayments. Areas of high chlorophyll *a* concentration in Yellowjacket Creek corresponded with high turbidity, suggesting that algal cells were a major source of light attenuation. Lowest chlorophyll *a* concentrations were in riverine areas of the lake and in open-water areas of the lower portion of the pool. Elevated inorganic turbidity in the riverine

reach of the lake and nutrient depletion in the lower portion of the pool may account for these observations (Kennedy, Thornton, and Gunkel 1982; Kennedy, Gunkel, and Carlisle 1983).

Cluster analyses, performed using only temperature, turbidity, and total pigment data, allowed identification of nine groups of similar stations. For the most part, stations contained in each cluster were also geographically related. An advectively ordered series of four clusters was identified for the upstream area of the lake from the inflow to the confluence of Yellowjacket Creek. Remaining clusters contained stations from the Yellowjacket Creek, Wehadkee Creek, and Whitewater Creek embayments and from downstream areas of the lake.

Results of comparison tests using supplemental water quality data (Table 2) provide additional characterization of station groups. Stations located in upstream areas of the pool (clusters 1-5) tend to experience reduced light availability due to the influences of nonalgal turbidity. While these locations are nutrient-rich, N:P ratios below 27 for many stations suggest the potential for nitrogen limitation (assuming favorable light conditions). Stations in clusters 4, 6, 7, and 9, many of which are located in the Wehadkee and Yellowjacket Creek embayments, have high chlorophyll *a*:total phosphorus and N:P ratios, suggesting that algal production is tightly linked to phosphorus supply. Changes in phosphorus loading to these areas would have a potentially greater impact than changes to turbidity-dominated areas of the lake.

Conclusion

Remote sensing provides a means for lake managers to identify complex spatial patterns in selected water quality responses. Evaluation of these patterns and their temporal persistence leads to increased understanding of relations between land use and lake response. This understanding provides an informational basis for developing and implementing water quality management plans.

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Table 2
Median Values and Groupings of Clusters Based on Comparisons of Selected Water Quality Variables

Variable/ Parameter	Cluster[1]								
	1	2	3	4	5	6	7	8	9
Total phosphorus, mg/L									
Median	0.151	0.148	0.094	0.051	0.093	0.028	0.021	0.081	0.041
n	5	10	39	14	55	109	191	6	114
Duncan group	A	BCD	B	CD	BC	D	D	CD	CD
Total nitrogen, mg/L									
Median	1.19	1.76	1.04	0.785	1.07	0.700	0.570	0.910	0.810
n	5	10	39	14	55	109	191	6	114
Duncan group	B	A	B	CD	B	CD	D	BC	BC
Secchi disk, m									
Median	0.600	0.400	0.450	0.600	0.800	1.40	1.50	0.800	1.20
n	5	10	39	14	55	109	191	6	114
Duncan group	CD	D	D	D	BC	A	A	B	A
Nonalgal turbidity, 1/m									
Median	1.70	2.18	2.31	1.19	1.06	0.313	0.384	0.768	0.321
n	5	10	39	14	55	109	191	6	114
Duncan group	AB	A	A	B	C	D	D	C	D

(Continued)

[1] Clusters with the same Duncan group identification have similar characteristics.

Table 2 (Concluded)

Variable/ Parameter	Cluster								
	1	2	3	4	5	6	7	8	9
Chlorophyll *a*, μg/L									
Median	1.41	2.58	5.42	19.0	15.2	16.0	12.2	18.0	20.0
n	5	10	39	14	55	109	191	6	114
Duncan group	F	E	D	AB	BC	ABC	C	AB	A
Chl*a*:TP ratio									
Median	0.009	0.013	0.034	0.474	0.075	0.594	0.610	0.206	0.521
n	5	10	39	14	55	109	191	6	114
Duncan group	D	D	D	AB	CD	A	A	BC	A
N:P ratio									
Median	23.8	27.8	21.8	33.4	25.0	51.9	59.4	34.1	46.7
n	5	10	39	14	55	109	191	6	114
Duncan group	D	ABCD	CD	ABC	BCD	A	AB	ABC	AB

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A Water Quality Program for Practical Engineers Under Strong Funding Constraints

by
E. Morton Markowitz[1]

Mathematics

The language we speak, whether English, French, or Spanish, depends heavily upon the speaker's knowledge of the language and that of the audience. The great value of a well-developed language lies in facilitating the communication of ideas clearly and accurately. In the scientific fields, even this isn't enough, and the languages of mathematics become a necessity. There are many different mathematical languages which are so abstruse that even highly trained mathematicians are not familiar with them all. Examples of those that are important to water quality specialists, as well as to most other engineers, are

Calculus and differential equations
Number theory, especially complex numbers
Boolean algebra
Vector analysis
Geometry
Probability theory and statistics

These are all fields of mathematics with which a competent hydrologist specializing in water quality must become familiar. This does not mean that a water quality specialist must know every theorem and all their applications, but as with English or French, a competent water quality hydrologist must be fluent in the mathematical languages needed for the job. Under strong funding constraints, very little of these mathematical fields can be taught as part of on-the-job training, and they must be learned outside working hours. The field of geometry referred to above will be used in an example of reservoir analysis later in this paper to show the importance of geometry and to apply geometry to a cost-effective method for developing reconnaissance-level hydrologic modeling.

We are out of school now, and we must be accurate in the work we do. Our work should be reliable (in the statistical sense of being both accurate and precise), error-free and without blunders, and, at the same time, cost effective.

Classical Mechanics

The basis for hydraulics and hydrology, including the field of water quality, lies in the field of classical mechanics; in the field of thermodynamics, which followed in the 19th and 20th centuries; and in the field of environmental engineering, which arose in the 20th century. From my own experience there are about 10 rules in classical mechanics with which every

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water quality specialist should become conversant. These principles are described briefly in the following paragraphs.[1]

Conservation of Matter

This principle, which was understood by the Greeks, was first clearly formulated by Antoine Lavoisier in 1774; it states that matter can neither be destroyed nor created, although the form of the matter involved in the reaction may be altered. The law was carefully tested by Landolt (1906) and found to be true within the limits imposed by experimental manipulation, i.e., 1 part in 10 million for all reactions that he studied. However, emission of radiation is accompanied by a loss of mass, equal to E/c^2, in which E is the energy of radiation and c, the velocity of light. To account for the conversion of mass to radiation, there is a law of conservation of mass and energy considered together; that is, the sum total of mass and energy remains the same in an isolated system.

Conservation of Energy

In a system of constant mass, energy can neither be created nor destroyed. Energy can be converted from one form to another, but it cannot be annihilated. Hermann von Helmholtz (1847) enunciated this principle in the following form: "In all processes occurring in an isolated system, the energy of the system remains constant." Energy and mass (see paragraph above) are to some extent interconvertible.

First Law of Thermodynamics

This principle is the logical consequence of the law of the Conservation of Energy, and states that mechanical energy and heat energy are *quantitatively interconvertible*. The change of energy of a thermodynamic system is equal to the heat transferred minus the work done. The validity of the law was tested by James Prescott Joule, who showed that $E = JQ$, where E is the work done in producing a quantity of heat Q, while J is a constant, known as Joule's Equivalent, and equal to 4.18 ergs per calorie at 15 °C.

Second Law of Thermodynamics

This principle states that "It is impossible for a self-acting machine, unaided by any external agency, to transfer heat from a body at a lower temperature to one at a higher temperature," or alternatively, "Heat cannot of itself pass from a colder to a warmer body." This means that if heat is to be transferred from a colder to a warmer body, work must be done by some external agency. Furthermore, no cyclic process is possible in which heat is absorbed from a reservoir at a single temperature and converted completely into mechanical work.

Conservation of (Linear) Momentum

This is the principle that, if a system of masses is subject to only internal forces that the masses of the system exert on one another, the total vector momentum of the system remains constant, and is equal to the sum of the product of the individual masses and their respective velocities; that is, for any array of several objects, the total momentum is the sum of the

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[1] Information derived from standard sources, including encyclopedias.

individual momenta. It is equivalent to the force required to bring the objects to a stop in a unit duration of time. Since momentum is a vector, the momenta of objects going in opposite directions can cancel to yield an overall sum of zero. The law of conservation of momentum is abundantly confirmed by experiment and can even be mathematically deduced on the reasonable presumption that space is uniform; that is, that there is nothing in the laws of nature that singles out one position in space as peculiar if compared to any other.

Conservation of Angular Momentum

If the sum of the torques resulting from external forces about a fixed axis is zero, then the total angular momentum of the aggregate of particles about that axis is constant. Both torque and angular momentum about an axis are vector quantities associated with rotational motion of a mass and are related to Newton's Second Law. This principle describes rotational motion in essentially the same way that ordinary momentum describes linear motion. Although the precise mathematical expression of this law is somewhat more involved than in the case of linear momentum, examples of it are numerous, and the principle has been thoroughly established by experiment. The principle can also be mathematically deduced by the same reasoning as linear momentum. The principle of the Conservation of Angular Momentum can be shown to follow mathematically from the reasonable presumption that space is uniform with respect to orientation.

First Law of Motion

This principle (also known as Newton's First Law and Galileo's Law of Inertia) was first enunciated by Galileo Galilei. The principle states that a body will remain at rest or in motion with a constant velocity unless an external force acts on the body. Said another way, the total linear momentum of a system (or body) is constant provided the system is not acted upon by an external force.

Second Law of Motion

This principle (also called Newton's Second Law) was first stated by Sir Isaac Newton. The principle asserts that the sum of the forces acting on a body is equal to the product of the mass of the body and the acceleration produced by the forces, with motion in the direction of the resultant of the forces. Said another way, the time rate of change of momentum is proportional to the force acting on the body. The dimensions of linear momentum (a vector) are force (also a vector) times time. It follows that, if a constant force acts on a body for a given time, the product of force and the time interval (the impulse) is equal to the change in the momentum. Conversely, the momentum of a body is a measure of the time required for a constant force to bring it to rest.

Third Law of Motion

This principle (Newton's Third Law) states that for every force acting on a body, the body exerts a force having equal magnitude and the opposite direction along the same line of action as the original force. This means that internal forces between particles exist in equal and opposite pairs, so that the sum of their components in any direction must vanish. If the total external force (or its component in any direction) is zero, then the final system obtained implies that the total linear momentum (or its component in that direction) will remain constant. This is known as the principle of **Conservation of Linear Momentum**. If two bodies

collide, the sum of their linear momenta remains constant. This is so because their interaction can only cause equal and opposite forces to act between pairs of particles. These forces, being internal to the system comprising the two bodies, can therefore produce no increment of total momentum.

Dimensional Analysis

The independence of a physical law from the particular system of units employed (also called completeness) is expressed by the so-called **pi** theorem, often attributed to Edgar Buckingham, a U.S. physicist. The physical law is thus demonstrably independent of any particular system of measurement, a statement previously merely held plausible. An important result obtainable from the **pi** theorem is the so-called principle of dimensional homogeneity, often referred to as the **Fundamental Theorem of Dimensional Analysis**. Dimensional analysis yields an amount of information dependent on the skill and experience of the analyst. Positive errors are never introduced by failure to recognize special situations; it is only that less than the maximum information is obtained.

Other Considerations

One of the "laws" most frequently overlooked is that of dimensional homogeneity (part of the fundamental theorem of dimensional analysis). You are probably aware that Manning's Formula is not dimensionally correct, and neither is the broad-crested weir formula. We frequently try to allay our consciences by assigning "dimensions" to the empirical constants in these formulas. Conversely, the often-scorned "rational formula" is dimensionally correct. In water quality we must try to prevent this state of affairs from becoming the rule and not the exception.

Moreover, few hydrologists even worry about thermodynamics; fortunately, however, most who do are in the field of water quality. Thermodynamics has its own rules that must be followed carefully.

Nothing has been said so far of biological or environmental engineering, of which we are becoming more aware as engineers, hydrologists, water quality specialists, and Corps of Engineers personnel. These fields too must be studied in school before coming to terms with your job. It is not my field, and I defer to others for a discussion of the relative importance of each facet.

Computers

Probably the most significant advance in hydrology and the study of water quality came about because of the development of computers in the 1960s. This made possible the development of sophisticated mathematical models in hydraulics, hydrology, and water quality, such as HEC-1, HEC-2, HEC-5, and HEC-5Q. Computers are not known as much for breaking new ground in hydrologic sciences such as water quality as they are for making it possible to apply a large number of data to the solution of practical problems and to calibrate mathematical models. At present, the major constraints appear to be in the kind and quality of the data amassed and the sheer number of the data. In short, the problems of the past which concerned the art of putting models together for analysis and systems development no longer appear to be the significant problems of the future. Future problems lie in the fields of data management: storage, retrieval, and data presentation. Storage is by far the most

pressing problem. We still do not know if we are obtaining all that we should, or whether we are obtaining the data in the right way.

Let me give a hypothetical example. In the South Pacific Division (including its Districts), we have been collecting water quality data for 20 years or so. We usually collect the data twice a year and record most of the samples in handwritten files for later retrieval and analysis. Thought was later given to the diurnal cycle, and what may be a high or low dissolved oxygen (DO) level compared to a previous year (same date). This year's high DO could have been the result of the time of day the sample was taken, as well as many other unrecorded factors. We will probably never be able to find out. Moreover, because of the sheer volume of records, we must give thought to storing the data in a sophisticated database management file, and being able to retrieve and analyze the data whenever such an idea crosses our minds; to do otherwise would condemn us to obtaining the same kinds of inadequate data for years to come.

Furthermore, with each passing year more and more of the records are lost. Despite their faults, these records may be the most valuable because they identify trends before they become problems or before they became Corps issues. This brings up the most challenging and pressing problem of our time: to develop a storage and retrieval system (data bank) that can be used for years without fear of going out of style, that can be related to graphical presentation and frequency distribution software, and can be presented in a variety of tabular formats to be set by the user. It must be contained in a system that is easy to back up (on floppy disks for example), safe and easy to transfer from computer to computer, and does not cost much to run or maintain. The data must be in suitable form for re-entry into future software programs. Most important, it is essential that this storage and retrieval system be developed as quickly as possible.

Modeling

Modeling falls into several different categories: mathematical (including digital), physical, analog, and graphical (including geometric). In 1955, when I first learned the method of characteristics (a complicated graphical or analytic method for flood routing under changing upstream and downstream mass and energy conditions), I recognized the need for researchers to employ electrical analog models, which were just beginning to be developed. The method of characteristics was just too difficult to apply. Early electric analogs had their own faults, the most important of which were the difficulty in transferring applications to other geometries, and the difficulty in slowing the model down to real time. It took a while until the construction of complex digital computers made it possible to solve similar problems easily.

In a parallel development--even earlier than the modern application of the method of characteristics--physical hydraulic models were being built for one purpose or the other. The major one I witnessed from the design stage on (from 1955) was the development of the San Francisco Bay Model. All modeling is expensive, and that model was especially so. Lately, the Bay Model has been used for salinity studies, but as in all distorted models, the difficulty in resolving Froude's number and Reynold's number for a given set of runs diminishes its acceptability, and allows for disagreement to enter from the outside with every alternative water quality application developed. Virtually all physical hydraulic and water quality models have problems with high cost and questionable accuracy, and I believe that we have to look again at the use of computerized digital modeling as a more credible and less expensive alternative to physical models. Dr. H. A. Einstein explained that to me many years ago, and it

took 25 years to see a way out of the dilemma. This came about through the development of modern computers and software.

Hydrologic Methods

We have become so enamored with the hydrologic techniques already developed that we are not paying attention to verifying the results we obtain, or making an honest report on the verification. There is no method we use that is so perfect that it should not be continually under review.

One method of verification that should be used much more often is the regional stream-study approach. This method uses independent parameters and a least-squares statistical determination to compute an expected dependent variable. Most often it is used in conjunction with a frequency statistic, such as the 50 percent-chance peak flow for the dependent variable, and drainage area and mean annual basinwide precipitation as independent variables. Another advantage of this method is that it is an inexpensive way to develop reconnaissance hydrology. Some of my results in the past have been estimated to lie within 10 percent of the correct answer 97 percent of the time, and I had no hesitancy in using the results for final design. Such methods can and should be developed for water quality studies.

Another method is the enveloping-curve approach. By taking maximum values of peak discharges at all sites and relating them to drainage area, for example, a curve of maximum values can be developed above which it is very unlikely that the "peak flows" at a particular site on a given stream could be exceeded. This kind of thinking can have a fountainhead of applications in the field of water quality.

Abbreviated Methods

The last section on hydrologic methods leads to another way of thinking about water quality. We cannot spend $1 million on developing the hydrology for a $2 million project. In looking at this problem, the Hydrologic Engineering Center in 1986 addressed the problem of sensitivity of water surface profiles to inaccurate basic data.[1] Similarly, we must study the problems of water quality with a view toward reducing the cost of the studies and still providing the planning and operations documents with water quality analyses of acceptable quality. Two abbreviated methods are presented in the following examples.

a. Example 1. In 1964 I observed the striking similarity between a number of reservoir stage versus storage curves: the shapes of the curves seemed to be about the same even though the absolute values varied considerably. I also noticed a correspondent similarity between reservoir stage versus surface area curves. With my trusty 2H pencil I drew some stylized geometric shapes, depicted in Figure 1. The equations of each (for area of base and for volume) are also shown. Turned such that the base is upward, these figures can represent lakes or reservoirs. I tabulated the results and saw that this was an application of Green's Theorem. My colleague at that time, Walter Sykes, saw that the relation between K and *alpha* (written herein as a lowercase a) was a simple formula

[1] US Army Engineer Hydrologic Engineering Center. 1986. Accuracy of computed water surface profiles. Research Document 26. Davis, CA.

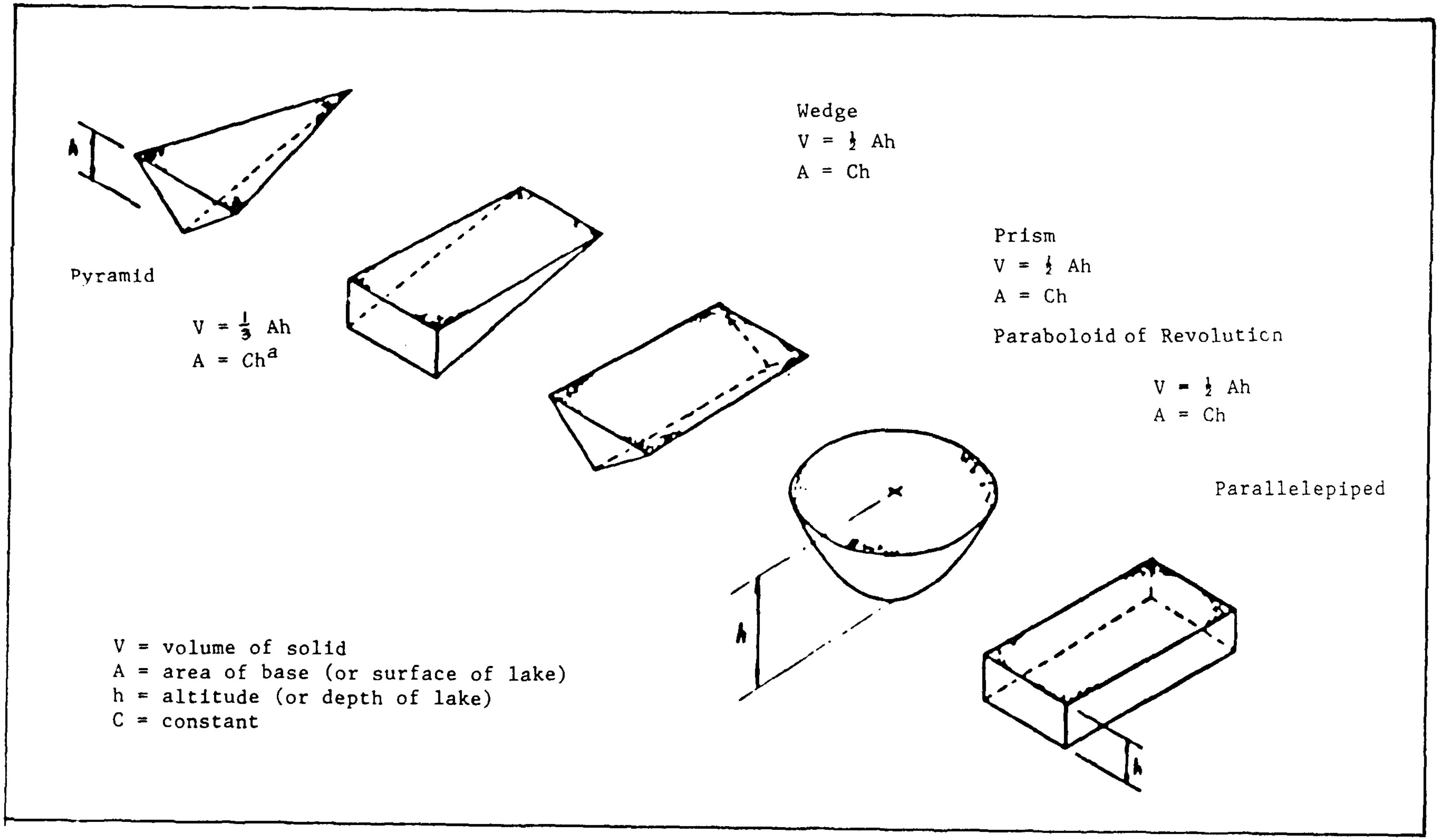

Figure 1. Characteristics of geometric solids

$$K = \frac{1}{1 + a}$$

That led directly to Figure 2. With the two basic formulas,

$$V = KAh$$

and

$$A = Ch^{a}$$

one can see that a natural lake or reservoir can be fit into the stylized pattern of a lake with very little loss of accuracy by assigning an appropriate K and a. The only thing left to do is establish the coefficient C, which is obtained by planimetering the surface area of the proposed reservoir at one elevation near the crest of the emergency spillway and noting the associated depth of the lake for that stage.

b. Example 2, Mass curves of runoff. This abbreviated hydrologic procedure has been around for a long time and should become standard fare for practicing hydrologists. Its usefulness stems from the need for reservoir volumes at different frequencies for such studies as dilution of toxic substances into reservoirs during floods. In simplified form the procedure is as follows:

(1) Develop frequency curves of runoff for several specified durations. Usually these curves are developed for periods of 1-, 2-, 3-, 5-, 10-, and 30-days duration. If they can also be developed for 12-hr duration without undue effort, so much the better. Peak flow versus frequency curves are very useful in this kind of analysis. Most methods for developing these frequency curves are acceptable, but runoff frequency curves developed directly from rainfall are suspect.

(2) For a given frequency, plot the 1-, 2-, 3-, 5-, 10-, and 30-day values on Cartesian (linear) coordinate paper against their volumes in cfs-days. Through the zero-duration point draw a line with a slope equal to the peak flow in cubic feet per second (i.e., cfs-days/day) at that frequency. Bearing in mind that the 1-day runoff volume is substantially less than the 24-hr volume or the 1,440-min volume, draw a smooth line through the points and asymptotic to the peak flow line to prepare the mass curve of runoff for a given frequency. This should be done for several frequencies in the neighborhood of the level of interest.

(3) Find the maximum allowable outflow in cubic feet per second. The maximum outflow is usually equal to the with-project downstream channel capacity. Take 0.7 of the maximum as the average outflow and plot it as a straight line superimposed on the mass curves of runoff and intersecting at the origin. The maximum ordinate distance between the mass curve of runoff for a given frequency

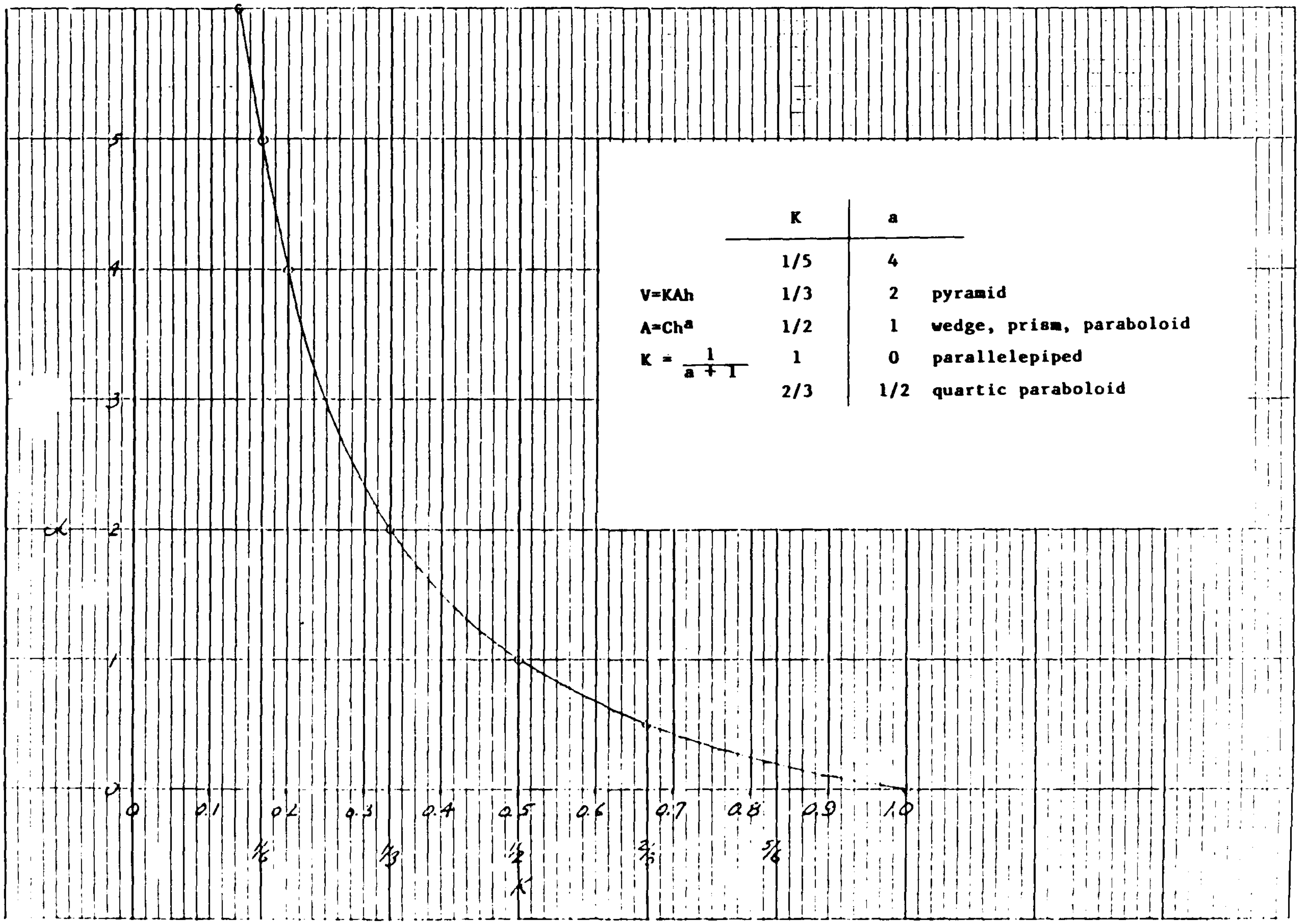

Figure 2. Application of basic formulas

and the average outflow will give the volume of reservoir storage required for a given flood frequency event.

The procedure is fast, simple, and reasonably accurate. It compares favorably to hydrograph analysis and has considerable value in developing drawdown curves needed for dam design. It is eminently suitable to feasibility study preparation. This author does not understand why such a valuable abbreviated method is so little used.

Computers and Communication

Something must be said here about networking and the transfer of information. Three major problems that are associated in communications with other computers must be addressed in the early future: viruses, breakdowns, and general costs.

a. Computer viruses. If you have not been victimized by a virus, consider yourself fortunate. Viruses are here to stay and are putting a serious burden on people who must use computers to transfer information to and from remote terminals. We can never be sure whether the output of our own computers truly records the output from the software interrogated nor can we rely on the calculations transferred. A whole field of computer science must be developed to keep this menace from forcing us to minimize or even abandon communication to and from "the outside."

b. Breakdowns. With the growing use of networking and the more efficient use of hardware, there has been a considerable increase in frustration due to the bottlenecks and breakdowns. For example, if a printer goes down, it goes out of service to a lot of people. Occasionally, other pieces in line (such as servers) break down or otherwise go out of service. The cost-effectiveness of savings in hardware is offset by costly installation, additional hardware, costly breakdowns and maintenance, increased dangers of viral infections, and increased layers of bureaucracy. All these costs and time delays must be compared with the costs of more local printers (personal printers), other local hardware, and some items of minor cost such as software.

c. General costs. Assume that 90 percent of the normal engineering workload can be accomplished on a standard 286 chip computer. A question is raised whether a computer should be purchased for the average workload. It would appear that the size of a computer purchase should be based on a reasonable maximum workload, very much like a dam and river system should be designed for a very rare event. Computers are expensive, and the costs must be judged in an analysis that includes this kind of thinking. This is particularly true inasmuch as some programs already developed and distributed by HEC cannot be run on average RAM storage. Whatever is done in this regard, computers and software purchased must be compatible with everyone in the same shop or line of work. Would you believe floppy disk drives incompatible with the disk drives of your secretary's computer?

Observations and Predictions

There is so much to be learned in the field of water quality that we cannot afford to train specialists in higher mathematics, general classical mechanics, and fundamental environmental engineering. These fields must be learned outside the job environment, usually before the

water quality specialist leaves school. I predict that, in part, this will become part of the job announcement.

I predict that more attention will be paid to classical mechanics in the work we do. Statistical and graphical formulas will fall farther behind in our final design reports, which will use thermodynamics, dimensional homogeneity, and greater verification of the methodology and results. It will be a challenge for the Corps to perform this under strong funding constraints.

The way we use computers is undergoing a rapid change from the days during which they were employed as word processors and as calculators for standard programs. They are being used more and more for analyzing data and for collecting data. They are being used to store data and print out finished reports. We must adjust to these changes. If we do, we will more effectively do our jobs under severe funding constraints than we ever could before.

I predict that digital hydraulic models will be the wave of the future; physical models are on the way out; geometric models will be incorporated into reconnaissance and feasibility studies; and analog models will have only limited use, probably in repetitive circumstances requiring reliable verification.

I am disturbed by the lack of imagination used to solve water quality problems. In the near future I hope to see a number of abbreviated methods for solving these problems in the same way that abbreviated methods in the past have been used to solve other hydrologic problems.

Although some experts in computers disagree with this, I believe that networking has been overused; that computer viruses are epidemic; and that costs and time delays are increasing unreasonably. As water quality specialists, we must rise to the occasion and meet this computer challenge in a fiscal environment beset with strong funding constraints.

Reaeration at Low-Head Hydraulic Structures

Steven C. Wilhelms,[1] John S. Gulliver,[2]
and Kenneth Parkhill[2]

Background

Presently, the most-cited water quality parameter in our rivers, lakes, and reservoirs is dissolved oxygen (DO). Oxygen concentration is a prime indicator of the quality of that water for human use and use by aquatic biota. Many naturally occurring biological and chemical processes use oxygen, thereby diminishing the DO concentration in the water. The physical process of oxygen transfer or oxygen absorption from the atmosphere or air bubbles acts to replenish the used oxygen. This process is termed "reaeration."

Low-head hydraulic structures within the Corps of Engineers are generally associated with navigation projects. These structures are usually "run-of-the-river" and have the objective of maintaining a constant upstream pool elevation. The effect of the deeper, slower pools is to reduce oxygen transfer as compared to the open river. Biological and chemical oxygen demands may accumulate and concentrate in the impoundment and thereby degrade the DO concentration in the stored water (because of the excess demand compared to reaeration capability). Without sufficient reaeration, release of this water may pose an environmental and water quality concern.

Some hydraulic structures exhibit remarkable reaeration, while others do very little to increase DO. The design engineer is faced with the need to evaluate the oxygen transfer characteristics of existing conditions at a hydraulic structure for comparison with the characteristics of a proposed modification. As the first step, the existing conditions must be understood, and transfer characteristics must be quantified.

The objective of this paper is to report results of an extensive review of predictive models for a variety of low-head structures. Based on reported data, predictions from several models were evaluated based on the uncertainty of their predictions. The "best" prediction models are recommended for application to "generic" hydraulic structures.

It is hoped that the field engineer, armed with the mathematical description of oxygen transfer at a "generic" type of structure and an understanding of how hydraulic conditions contribute to oxygen transfer, should be able to estimate the reaeration that is occurring at a specific structure and "bracket" the uncertainty of that estimate.

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Reaeration Description

In turbulent flows, reaeration is usually considered to be a first-order process in which the rate of change of DO concentration in the water is linearly dependent on the ambient concentration. The "saturation deficit" is the driving force in the transfer process and is the difference between the actual concentration of the oxygen in the water and the "saturation concentration" that corresponds to equilibrium with the air. A convenient parameter, called "transfer efficiency," can be defined as the fractional part of the incoming deficit that is satisfied as the water flows through the structure. This efficiency E can be expressed mathematically with

$$E = \frac{(C_d - C_u)}{(C_s - C_u)} = 1 - \frac{D_d}{D_u} = 1 - \exp\left[-k_L \frac{A}{V} t\right] \tag{1}$$

where

C_d , C_u , C_s = downstream, upstream, and saturation concentrations, respectively
D_d , D_u = downstream and upstream saturation deficits, respectively
k_L = liquid film coefficient
A = interfacial surface area for gas transfer
V = volume of water affected
t = contact time

If the transfer efficiency is zero, no oxygen transfer occurred, and the downstream concentration equals the upstream concentration. If the transfer efficiency equals 1.0, then all of the upstream deficit was met by the oxygen transfer and the downstream concentration is at saturation. From the perspective of flow through a hydraulic structure, the initial and final deficits would be the upstream and downstream deficits, respectively, and the elapsed time would be the time of flow from the upstream to downstream locations.

Important Physical Processes

Wilhelms and Gulliver (1990) identified three basic processes occurring at hydraulic structures that can significantly increase oxygen transfer. These processes are described below.

Turbulent mixing at the water surface and within the body of the flowing water significantly affects gas transfer because turbulence transports mass much faster than molecular diffusion. The diffusive layer (concentration boundary layer) at the air-water interface is kept thin by impinging turbulent eddies, causing increased gas transfer. A high degree of turbulent mixing, such as occurs on the face of a spillway or in a tailwater plunge pool, increases k_L.

Entrained air in the flow, either because of surface aeration or plunging flow, dramatically increases the interfacial surface area A.

Higher pressure, which bubbles experience in the tailwater plunge pool, increases the saturation concentration C_s in the bubbles and thereby increases the saturation deficit resulting in enhanced oxygen absorption.

By recognizing these processes and their impacts on oxygen transfer, flow conditions observed in a physical model or full-scale project can, at a very minimum, be qualitatively evaluated.

Hydraulic Structure Categories

Low-head structures range widely in purpose, size, and configuration. Consequently, the hydraulics are extremely varied. However, the American Society of Civil Engineers (1991) categorized open channel structures into four groups: (a) free-surface flows, such as flow in a channel or on a gated or ungated spillway or ogee crest (Figure 1); (b) submerged flows, such as discharge under a submerged gate (Figure 2); (c) free jets, such as flow over a sharp-crested weir (Figure 3); and (d) transitional flows, where free surface flows or jets interact with a pool of water resulting in plunging flow or a hydraulic jump (Figures 4 and 5). The hydraulics of each group differ greatly, resulting in significantly different gas transfer characteristics.

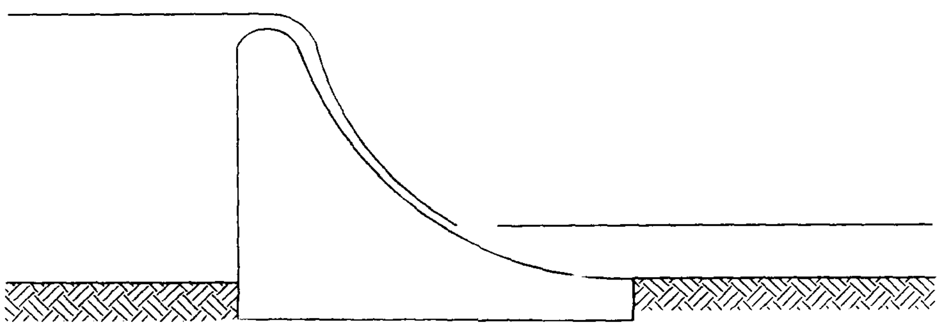

Figure 1. Gated or ungated ogee crest

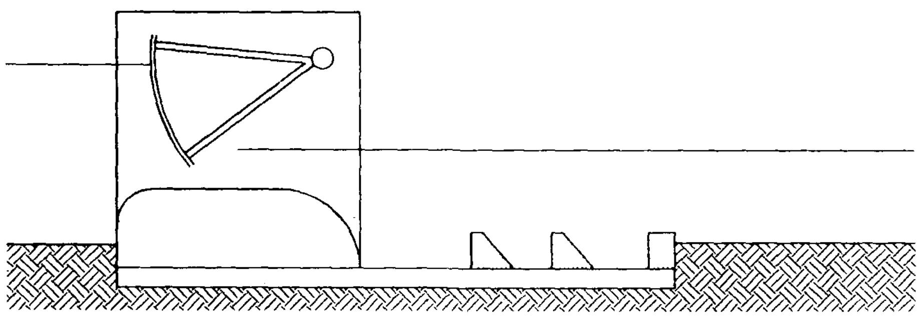

Figure 2. Submerged flow at gated sill

Predictive Equations

Predictive equations or mathematical models for reaeration at hydraulic structures generally use physical parameters of the structure or flow conditions, i.e., Froude number, depth of tailwater, discharge, etc., to estimate reaeration efficiency. Most equations are more or less empirical, with intuition being used to specify the independent variables and regressions used to determine the constants in the equation. In most instances, each equation is derived from a different data set, with coefficients adjusted to fit the specific data of the developer. Because

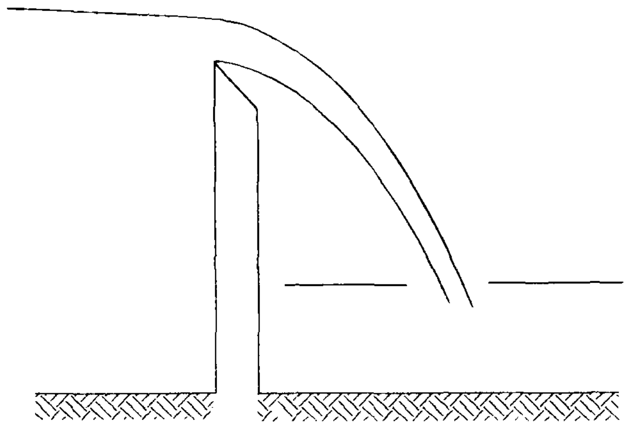

Figure 3. Spill over a sharp-crested weir

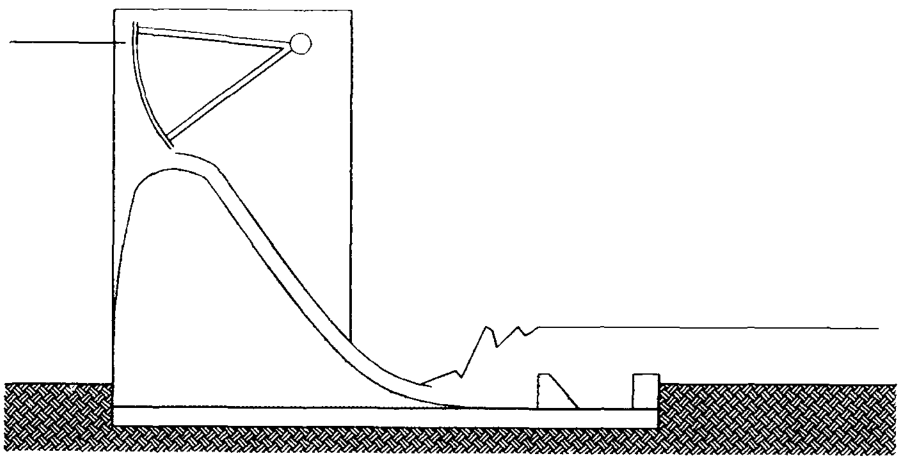

Figure 4. Flow at a spillway with hydraulic jump

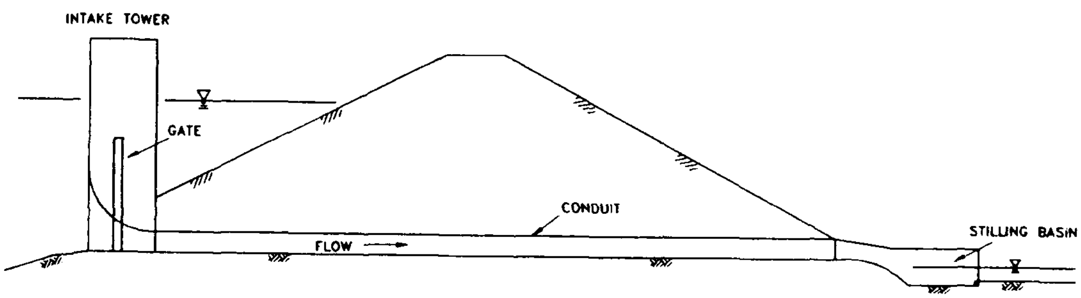

Figure 5. Flow through a gated conduit

the size of most datasets is relatively small, large deviations between predicted values of gas transfer efficiency and measured values at different sites are common. It should be recognized that predictive equations are reliable only as first approximations, and that they should be used carefully only when more accurate and detailed studies are not feasible.

The equations reported herein are those recommended by Wilhelms, Gulliver, and Parkhill (in preparation). The reader should refer to that report for a more thorough explanation of the development of these and the other equations investigated and their respective references. While an equation developed specifically for a project provides the best approximation for reaeration efficiencies at that site, the equations given below, which minimize uncertainty in predictions for generic structures, provide the best estimate for reaeration when a specific equation is not available.

Wilhelms, Gulliver, and Parkhill (in preparation) assembled and screened, through an uncertainty analysis, several sources of observed data. Predictions from 11 equations that describe gas transfer efficiency at various types of structures were compared to this comprehensive database. For gated and ungated ogee crests, Equation 2 (Rindels and Gulliver 1991) showed the smallest standard error of 16 percent.

$$E = 1 - \exp\left[\frac{-0.08H}{1 + 0.02q} - 0.062Z_p\right] \tag{2}$$

where

H = head across the structure, ft
q = unit discharge, cfs/ft
Z_p = tailwater depth, ft

For gated sills, Pruel and Holler's (1969) model (Equation 3) more closely predicted the observed values with a standard error of estimate of 14 percent.

$$E = 1 - \frac{1}{\left(1 + 666N_f^{-3.33}\right)} \tag{3}$$

where N_f is the Froude number at impact

$$N_f = \frac{(2gH)^{0.75}}{(gq)^{0.5}}$$

and g is the acceleration due to gravity. The transfer efficiency of flow over a sharp-crested weir was predicted more accurately by Avery and Novak's (1978) model

$$E = 1 - \left[\frac{1}{1 + 0.64 \times -4\ F_j^{1.787}\ R^{0.533}}\right]^{1.1149} \tag{4}$$

where F_j is the Froude number of the jet, given by

$$F_j = \frac{(2g)^{0.25}\ H^{0.75}}{q^{0.5}}$$

and R is the Reynolds number of the jet defined by

$$R = \frac{q}{2\nu}$$

where ν is the kinematic viscosity. Predictions resulted in a standard error of 17 percent. Gated conduit outlets demonstrate the transitional characteristics of several flow conditions. Equation 5 (Wilhelms and Smith 1981), with a standard error of 31 percent, is recommended for prediction of reaeration through this type of structure.

$$E = 1 - \exp\left(-0.045H\right) \tag{5}$$

Summary and Conclusions

Gas transfer or reaeration is considered a first-order process and can be conveniently described by Equation 1. The important physical processes that impact gas transfer are included in this formulation. The effect of turbulent mixing is reflected by the liquid film coefficient k_L. The impact of surface area available for gas transfer, which must include the surface area of entrained air bubbles, is represented by the specific area term A/V. Enhanced gas absorption due to the effects of hydrostatic pressure on plunging aerated flow is included as a pressure modification of the saturation concentration C_s.

Generally, low-head structures can be categorized into four groups: (a) free-surface flows, such as flow in a channel or on a spillway or ogee crest; (b) submerged flows, such as discharge under a submerged gate; (c) free jets, such as flow over a sharp-crested weir; and (d) transitional flows, where free surface flows or jets interact with a pool of water resulting in plunging flow or a hydraulic jump. The hydraulics of each group differ dramatically and, consequently, the gas transfer characteristics are significantly different. However, an understanding of the hydraulics and the resulting gas transfer characteristics of each type of structure can permit limited extrapolation to other hydraulic structures.

Results of an extensive database and predictive model evaluation show that Rindels and Gulliver's (1991) model (Equation 2) is most accurate for gated and ungated ogee spillways. Pruel and Holler's (1969) model (Equation 3) predicts reaeration at gated sills most accurately. The model (Equation 4) of Avery and Novak (1978) best estimates the transfer efficiency for weirs, while Equation 5, the model of Wilhelms and Smith (1981), best describes the reaeration of gated conduits.

In the efforts to predict the impact of hydraulic structures on levels of dissolved gases in river systems, the development of each equation was limited by the size of the database and the difficulty encountered in deriving an appropriate theory from first principles. As a result, many equations are useful only for specific types of structures under particular conditions. Relatively large errors may result if predictive equations are applied indiscriminately. Combining the use of the recommended models and specific, but limited, field measurements will provide the most consistent means of determining oxygen transfer characteristics.

Acknowledgment

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Development of an Expert Advisor for Selective Withdrawal Operations

by
J. P. Holland[1]

Introduction

Density stratification in lakes and reservoirs results from differential heating, primarily within the vertical strata of those water bodies. A consequence of this stratification is that differing thermal resources are found within those strata. The well-known concept of selective withdrawal is used at many of these impoundments to allow for the discharge of specific, and often rigid, release temperature requirements.

A primary concern in the use of selective withdrawal for release temperature maintenance involves the effective selection of which port, among several vertical withdrawal locations, should be operated for required temperature control. This selection process is of importance because resource limitations within most impoundments require the blending of waters from multiple impoundment strata to achieve required release temperature objectives. The process of selecting which withdrawal locations to operate often involves consideration of a large and competing constraint set made up of structural, hydraulic, and environmental concerns.

Selective Withdrawal Operational Considerations

A number of hydraulic, hydrologic, and structural concerns must be incorporated into any decision of which selective withdrawal ports to operate. These include consideration of the total discharge from the project for a given time period (usually 1 day), the maximum flow allowable through the selective withdrawal system, the minimum and maximum allowable flows through each of the withdrawal ports and the floodgates, the type of outlet structure being operated, and the number of available wet wells in said structure.

In addition, a number of environmental concerns must be integrated into port selection decisions. These include the required downstream target temperature, changes in these targets over a stratification season, seasonal requirements for fish passage from certain strata of the impoundment, and concerns relative to other water quality parameters such as dissolved oxygen, iron, and manganese. It should be noted that selective withdrawal operational decisions must also reflect proper consideration of other project purposes such as hydropower, recreation, water supply, navigation, and flood control.

A number of methods have been used to effect selective withdrawal port selection decisions. The method traditionally used is referred to as manual operation based on "engineering experience." This method usually involves the selection of withdrawal ports based on experience gained by field personnel in the operation of the given project over an extended period.

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This method, however, has several potential limitations: (a) it does not systematically consider the variety of constraints that must be assuaged by a single selective withdrawal operation; (b) since the method is based solely on experience of the operators, its worth decreases if release target temperatures change or the operators leave; (c) the method is extremely difficult to effect for single-wet well operations because of the impacts of the density stratification on the flow distribution between multilevel ports; and (d) as a result of the "manual" nature of this method, which often requires estimation and refinement of which ports should be operated (and how much flow should pass through them), valuable project thermal resources may be wasted.

Advisor for Selective Withdrawal Operations

To augment selective withdrawal operational decisions, a knowledge-based advisor has been developed. This advisor couples three specific components: (a) a numerical code that computes withdrawal distributions for a variety of outlet devices; (b) a port selection algorithm; and (c) an expert shell that couples the two above components with a number of heuristic operational rules. Each of these components is discussed below.

Withdrawal zone computations

Appropriate selective withdrawal operational decisions require that a decision-maker have the ability to accurately predict the vertical distribution of withdrawal for a given outlet device from a density-stratified impoundment, and then apply that knowledge to effective selection of withdrawal elevation/location. As an example, consider an outlet structure with a port relatively near the surface of the impoundment. It is reasonable to expect that said port will withdraw primarily surface water. However, for specific stratification and discharge conditions, the flow through that port may be made up of a significantly decreased volume of surface water. This problem is further complicated when ports reside in the metalimnetic strata of the impoundment. The consequence of inaccurately predicting the withdrawal distribution for a given port, and by inference the temperature entering said port, is often to badly miss the required downstream release temperature objective.

SELECT (Davis et al. 1987) is a one-dimensional numerical selective withdrawal code that implements analytical and experimental knowledge to predict vertical withdrawal zone distributions from a density-stratified impoundment. The code computes the withdrawal distribution based on user-specified outlet device type (port or weir), temperature or density profile, discharge, and other considerations. SELECT computes, based on the predicted withdrawal distribution and user-specified profiles for conservative water quality parameters, the release characteristics for those parameters. It should be noted that SELECT is not a water quality or thermal model; rather, its purpose is to compute withdrawal characteristics.

Port selection algorithm

The SELECT code requires that the user specify the location, hydraulic characteristics, and flow(s) through the port(s) being operated. Thus, through simulation of a number of scenarios, the user could converge to the appropriate combination of ports to operate to meet a release temperature requirement. To alleviate the need for such a procedure, a routine was coupled with the SELECT code to simulate the operation of a selective withdrawal structure. This routine, named DECIDE (Dortch and Holland 1984), requires as input the thermal

stratification of a given impoundment, the downstream temperature objective, and the total flow to be released for a given day or period. Based on these inputs, the hydraulic characteristics of the withdrawal structure, and operational heuristics defined as procedural rules, the routine determines the combination of selective withdrawal ports to be operated and the flows to be released through those ports, such that the released temperature is as close as possible to the downstream temperature objective.

The heuristic rules are used to limit the large number of combinations of ports and release flows from those ports that could potentially be used to achieve the release temperature target, thereby increasing the computational efficiency of the algorithm. These heuristics include rules that favor the use of a limited number of adjacent ports and the use of ports in different wet wells.

To compute the flows required through selected ports, the temperatures of the flows withdrawn through those ports must be accurately predicted. These values are provided by the SELECT code. At present, a dual-wet well version of DECIDE has been incorporated into the global advisor framework; a single-wet well version of the routine will be added in the near future.

Expert shell

The purpose of the expert shell (Waterman 1986) is to provide a dialog system interface that allows the user to easily operate the SELECT model. The interface was designed to be menu driven and can be used with a mouse. The dialog system was developed as a program-controlling shell to maintain the integrity of the original model. The shell is written in the C programming language because that language provides the ability for graphical display and control on personal computers and can access the executable version of the SELECT model (which is written in FORTRAN). The dialog interface is used to build and modify the input data files for the main computation module and to display the results from the computations. By operating at the level of the input and output files, the integrity of the original model and the format of the input and output files can be maintained.

The SELECT model requires basic input data that include a description of the number and location of selective withdrawal ports, the description of the wet wells and the configurations, details of flood control outlets, and hydraulic constraints in terms of minimum and maximum port flows. It also requires a description of the thermal and quality structures within the impoundment immediately upstream of the withdrawal structure. These data can be provided via the user interface, or the interface can be used to modify a previously created data file through an internal screen editor. One can also control whether the SELECT model will run in the mode in which the user supplies individual port releases or the mode in which the user provides a desired target temperature and the advisor determines the most appropriate port releases. For either mode, the model is run from within the shell.

Once the model is run, the user can display plots of the normalized withdrawal and density structure as a function of depth within the impoundment for any feasible port release strategy. Model results are available that summarize the amount of flow released from each port (both as a quantity and as a percentage of total structure release flow) and the computed release characteristics for up to four different release temperature targets. By using other menu options, the user can examine port operations for each individual temperature target.

The advisor also has a variety of window selections that can be made in terms of information to be displayed, and the windows can be moved and resized on the screen. The screen information can also be output to a printer. This structure allows the user to rapidly evaluate and compare port operation strategies. The user can then easily modify operational mode, target temperatures, port configurations, or the temperature data and run other scenarios. Since the original output file from the SELECT model is maintained, this file can also be printed or used as input to other graphics packages for postprocessing activities.

Inclusion of heuristic rules

Currently the heuristic rules are incorporated as procedural rules within the DECIDE subroutine. The procedural rules are invoked when the user chooses the mode of operation that allows the model to compute port operational strategies. This structure has the advantage that it allows development and expansion of the DECIDE subroutine within the context of the existing interface. Provision has been made in the dialog interface for future development where the user could potentially select from a menu of the most appropriate operational heuristics that could then be enforced for a given project. These rules would then be invoked by sending necessary input data values to the SELECT model.

The developed software package will run on IBM-compatible personal computers with automatic detection of the video capabilities. The model is currently being developed on 286- and 386-class personal computers with math coprocessors and EGA and VGA color graphics. The time required to run the SELECT model for each target is less than 5 sec on 386-class systems.

Conclusion

A rule-based computational advisor that aids in the selection of selective withdrawal operations for project release temperature maintenance has been presented. This advisor couples a one-dimensional numerical model that computes selective withdrawal vertical distributions in a density-stratified reservoir with a port selection algorithm and an expert system shell. Heuristic rules are built into the port selection algorithm presently, provisions having been made for incorporation of such rules from the expert shell in the future. The advisor is designed to run on personal computers to facilitate ease-of-use by project personnel.

The presented computational advisor for selective withdrawal operations is believed a useful and viable product that will aid field personnel in port selection decisions for release water quality maintenance. The advisor is of modular design, allowing for future modifications and site-specific customizing, and will run on most personal computers. The system is specifically developed to allow for systematic evaluation of the multitude of hydraulic, hydrologic, environmental, and structural considerations that accompany any port selection decision. Further refinement of the advisor, in the form primarily of incorporation of user-developed heuristic rules for specific projects, will continue.

Acknowledgment

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Analysis of Buoyant Flows in Density-Stratified Impoundments

by
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Abstract

Reservoir design and operation affect water quality both in the impounded pool and in project releases. In Corps of Engineers reservoirs, operational or structural features significantly influence the distribution of water quality characteristics. Operational and design influences may involve hydropower, spillway, flood tunnel or sluiceway releases, inflows resulting from pumpback or projects upstream, and internal flows generated by oxygenation or destratification systems. Traditional means of analyzing these flow characteristics involve field studies and/or physical models, which can be prohibitively expensive. One-dimensional numerical reservoir models have been used over the past 20 years to address impacts on vertically averaged reservoir water quality. These models cannot resolve the detailed near-field hydrodynamics resulting from project operation or structural modifications that affect project water quality. Advances in computational fluid mechanics have enabled the development of multidimensional numerical models that can address many of these flow phenomena. Applications will be presented involving the simulation of buoyant flows in density-stratified impoundments with important implications to project water quality.

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Economic Benefits of Fishery Improvements Associated with Lake Destratification

by
Richard E. Punnett[1] and Michael E. Hoeft[2]

Abstract

Lake destratification was conducted from 1987 to 1990 at Beech Fork Lake, a 740-acre lake in West Virginia, to improve the fishery habitat. Improvement in the habitat was associated with increases in standing crop, harvest, catch, and recreational use. The economic benefits of the improvements were evaluated and compared to the cost of the destratification program. The benefit-cost ratio ranged from 1.0 to 93.0 times the annual cost depending upon the fishery evaluation technique. On an annual basis, net benefits ranged from -$24 for the harvest valuation method, to $24,243 for the catch valuation method, to $95,582 for the fisherman use valuation method, to $529,373 for the standing crop valuation method.

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Nutritional Implications of Convective Hydraulic Circulation

by
John W. Barko[1] and William F. James[2]

Abstract

On a daily basis, shallow littoral regions of lacustrine systems heat and cool more rapidly than adjacent open-water regions, partly as the result of differences in mixed volume. Submersed aquatic vegetation can contribute significantly to the development of resulting thermal gradients, which promote convective hydraulic circulation. Effects of convective hydraulic circulation are potentially far-reaching, since this mechanism operates daily and can account for massive transport of dissolved constituents, including nutrients, between littoral and open-water regions of lakes and reservoirs. For example, hydraulic transport affected by convection has been demonstrated to account for about 26 percent of the internal load of phosphorus to a small Wisconsin reservoir. Ongoing studies of internal nutrient loading in reservoirs indicate important effects of convective circulation on lacustrine nutrient and phytoplankton dynamics.

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Sedimentation Dynamics in Eau Galle Reservoir, Wisconsin

by
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Introduction

The occurrence of greater sediment accumulation in deeper regions of lakes than in shallow regions has been largely attributed to erosional mechanisms that collectively result in sediment resuspension, redeposition, and focusing (Likens and Davis 1975, Håkanson 1977). Davis and Brubaker (1973) suggested that sediment is resuspended primarily from shallow sediments, mixed throughout the water column, and deposited uniformly over the lake basin during overturn events, resulting in focusing of shallow sediments to deeper regions. While this conceptual model explains the depth-related changes in both sediment accumulation and composition in lakes, and suggests that autumnal peaks in sediment trap rates in the profundal region are the result of sediment focusing, more evidence is needed both to support the hypothesis of Davis and Brubaker (1973) and to quantify sediment resuspension, redeposition, and focusing processes. We compare both sediment trap and sediment core rates at several depths in Eau Galle Reservoir, Wisconsin (see Barko et al. 1990 for reservoir description), to suggest hypotheses regarding regions of sediment resuspension, redeposition, and focusing and the origins of redistributed sediment. Based on these comparisons, we estimate rates of sediment resuspension, redeposition, and focusing during one period of autumnal overturn.

Methods

Sediment trap rates (ST rates) were measured at biweekly to monthly intervals from May 1986 to November 1987. The sediment traps were deployed 0.5 m above the bottom at the 2-m (two stations), 4-m (seven stations), 6-m (four stations), and 8-m (two stations) depths along transects that radiated from the center of the reservoir to the shoreline (Figure 1). The sediment traps consisted of three replicate cylinders, having aspect ratios of 5.0 (50-cm length/10-cm diameter) attached to a carousel frame. Carousels were held vertical in the water column by a buoy suspension system. Samples from cylinders were thoroughly homogenized and subsampled for dry mass. Dry mass was determined on glass fiber filters dried to a constant weight at 105 °C. Daily sediment trap rates were calculated as dry mass divided by the product of the cross-sectional area of the cylinder and days of deployment. Annual ST rates were calculated for the period April 1986-87 and November 1986-87 as the sum of daily sediment trap rates over each deployment period for an annum.

During the winters of 1986 and 1987, sediment cores were collected under ice near the sites of sediment trap deployment (Figure 1). Sediment cores were collected with a gravity core sampler (Wildco Wildlife Supply) equipped with plastic core liners (5.1-cm inside diameter and 91.4-m length). Each core sample was sectioned at 5-cm intervals until

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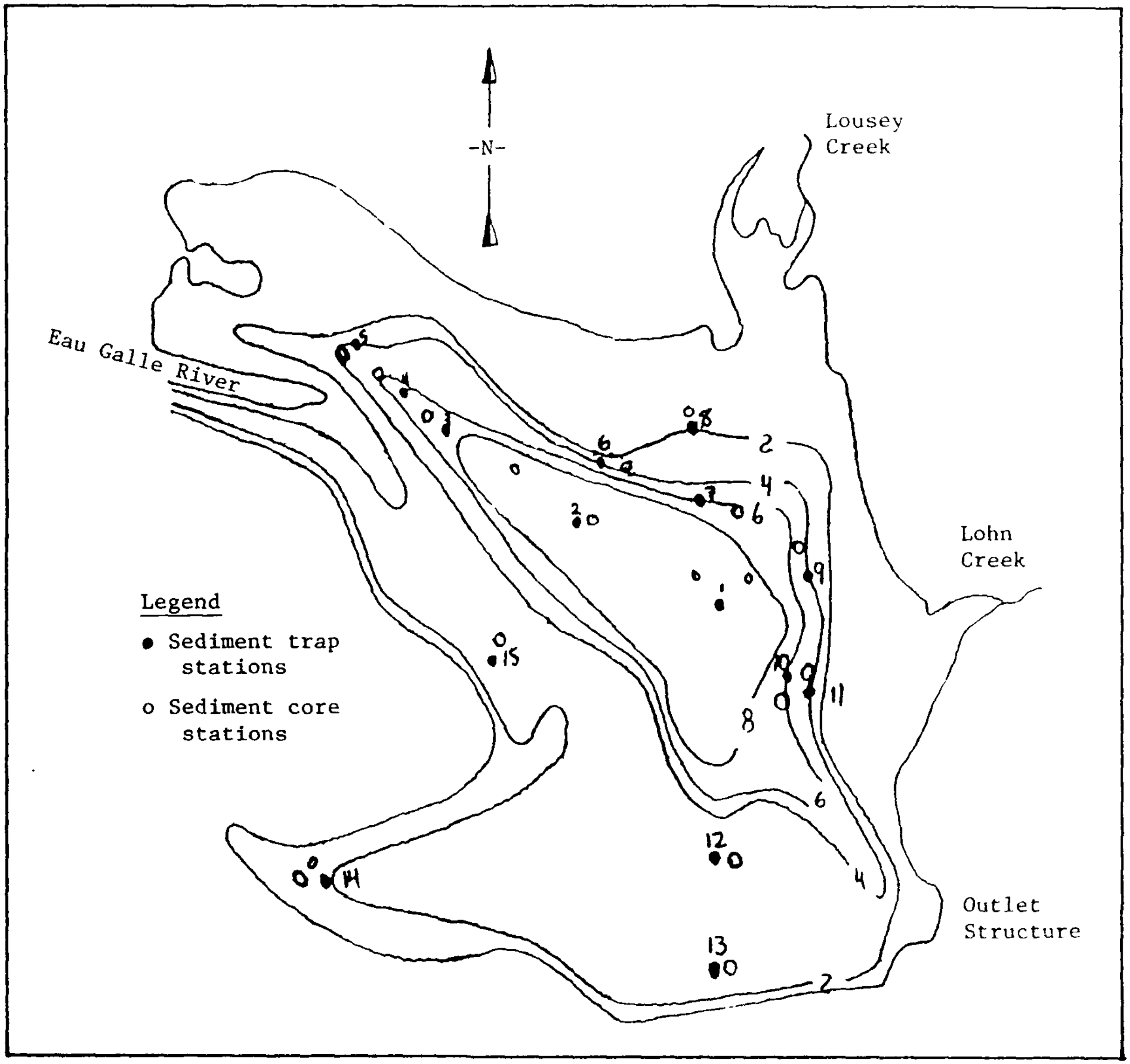

Figure 1. Morphometric contours (in meters) and sediment trap and sediment core stations in Eau Galle Reservoir

preimpoundment soils were encountered (i.e., James and Barko 1990). Core sections above the preimpoundment soil were dried at 105 °C for dry mass determination. Annual sediment core (SC) rates, representing an average sedimentation rate over 17 years, were calculated as the sum of dry mass above preimpoundment soils divided by the product of cross-sectional area of the core liner and reservoir age.

Water temperature was monitored weekly to biweekly at Station 1 (Figure 1) during a period of cooling and autumnal overturn in 1987 (August-November). Measurements were taken at 0.5-m intervals from the reservoir surface to the bottom. Air temperature and wind

speeds were monitored each hour during this period with a recording thermistor probe and anemometer (OmniData International).

Results

Mean annual ST and SC rates varied as a function of water depth (Table 1), with rates increasing with increasing depth at depths >4 m. Annual ST rates were significantly greater ($p < 0.05$; T-test) than annual SC rates at depths ≤4 m. In contrast, annual ST and SC rates were similar at depths >4 m. Reservoir-wide ST rates, integrated over sediment area, overestimated reservoir-wide SC rates by 0.9 kg m^{-2} $year^{-1}$ (Table 1).

Table 1
Comparison of Mean (±1 S.E.) Annual ST and SC Rates (kg m^{-2} $year^{-1}$) at Four Depth Strata in Eau Galle Reservoir

Method	Depth Stratum 2-m	4-m	6-m	8-m	Reservoir-wide
ST rate	3.5 (0.3)*	4.2 (0.2)*	6.1 (0.3)	9.6 (2.4)	5.5
n	4	12	8	4	
ANOVA	c	b,c	b	a	
SC rate	2.3 (0.2)	1.9 (0.6)	6.9 (0.8)	10.5 (0.9)	4.6
n	3	6	4	3	
ANOVA	c	c	b	a	
Area, m^2	144,405	229,783	107,631	121,184	603,003

Notes: Annual ST rates were calculated for the years April 1986-87 and November 1986-87. Asterisks represent significant differences between ST and SC rates ($P < 0.05$; T-test). Different letters indicate significant differences at the 5-percent level or less (Duncan's Multiple Range Analysis) between depths. Reservoir-wide ST and SC rates were calculated as the sum of rates integrated over areas enclosed by each depth stratum.

Since sediment resuspension, redeposition, and focusing are likely to occur during periods of overturn, we examined ST rates and weather patterns during the autumn overturn period of 1987, in an effort to account for some of the discrepancy between annual ST and SC rates (Table 1). During this period of overturn, water temperatures cooled and stratification weakened during August through September, coinciding with steady decreases in mean daily air temperature, and isothermal conditions existed by late September. Increases in mean daily wind speed and decreases in mean daily air temperature occurred during October. Water column temperatures cooled rapidly and remained isothermal during October through November.

During the summer stratified period, time-integrated daily ST rates were similar at 2 and 4 m, and exhibited slight increases with increasing depth at depths >4 m (Table 2). In

Table 2
Comparison of Time-Integrated Daily Summer (1 June-30 September) Versus Daily Autumnal (1 October-30 November) ST Rates at Four Depth Strata in 1987

Period	Depth Stratum			
	2-m	4-m	6-m	8-m
Summer	11.2	10.3	15.6	19.8
Autumn	27.8	27.7	38.9	51.0

Note: Daily ST rates were integrated over time to obtain an average daily rate.

contrast, time-integrated daily ST rates increased substantially at all depths during autumnal overturn (i.e., 1 October-30 November; Table 2). Increases over the summer period were greatest at depths >4 m.

A conceptual framework, similar to the one suggested by Davis and Brubaker (1973), was developed to interpret these sediment trap patterns during autumn overturn, and to estimate sediment resuspension, redeposition, and focusing. In our conceptual framework, we assumed that sediment resuspension occurs primarily at depths ≤ 4 m during autumn overturn, and that a portion of this resuspended sediment is redeposited back to depths ≤ 4 m while a portion is focused to depths >4 m. Thus, increases in daily ST rates at depths ≤ 4 m during autumn overturn (Table 2) were the result of sediment redeposition, because annual ST rates significantly overestimated annual SC rates at these depths (Table 1). Increases in daily ST rates at depths >4 m during autumnal overturn were assumed to result from sediment focusing only, and not from sediment resuspension and redeposition at depths >4 m, because annual ST rates were similar to annual SC rates at these depths (Table 1). If sediment resuspension and redeposition were occurring at depths >4 m, we would have expected annual ST rates to be greater than annual SC rates. For all these deposition processes to occur, it was further assumed that homogeneous mixing of resuspended sediment particles occurred throughout the water column.

We estimated apparent sediment focusing during autumnal overturn of 1987 as the difference between integrated daily summer and autumnal ST rates (i.e., baseline method of Håkanson) at depths >4 m (Table 2). The rate differences, representing focused sediment, were then integrated over reservoir areas below 4 m (e.g., rate differences at 4 m $\times$ area of the 4- to 6-m depth stratum plus rate differences at 8 m $\times$ area of the 6- to 8-m depth stratum), multiplied by the estimated length of autumn overturn (i.e., 61 days, 1 October to 30 November), and normalized with respect to reservoir surface area (0.6 km^2) to calculate the mass of sediment focused during the entire autumn overturn period. The apparent sediment focusing rate of 0.6 kg m^{-2} $year^{-1}$ during autumn overturn accounted for 19 percent of the annual SC rate at depths >4 m (Table 1).

Apparent sediment redeposition during autumnal overturn was similarly estimated using differences between daily summer and autumnal ST rates for depths ≤ 4 m (Table 2), normalized with respect to reservoir surface area. The apparent sediment redeposition rate of 0.6 kg m^{-2} $year^{-1}$ represented 46 percent of the annual SC rate at depths ≤ 4 m. Apparent sediment

resuspension during autumn overturn was calculated as the sum of the sediment focusing and redeposition rates. The apparent sediment resuspension rate of 1.2 kg m^{-2} $year^{-1}$ accounted for over 90 percent of the annual SC rate at depths ≤4 m.

We corrected reservoir-wide annual ST rates (Table 1) for the sediment resuspension rate during autumn overturn for direct comparison with reservoir-wide annual SC rates. The sediment resuspension rate was used to account for (a) sediment collected by sediment traps during quiescent periods (i.e., summer stratified period) that was later focused during autumnal overturn and (b) sediment redeposited into sediment traps during autumn overturn. The corrected reservoir-wide annual ST rate of 4.3 kg m^{-2} $year^{-1}$ was much closer to the reservoir-wide annual SC rate of 4.6 kg m^{-2} $year^{-1}$, suggesting that sediment resuspension at depths ≤4 m during autumn overturn explained a large portion of the discrepancy between annual ST and SC rates.

Discussion

Analyses of daily ST rates and annual ST and SC rates allowed for hypotheses regarding the delineation of regions of apparent sediment resuspension, redeposition, and focusing in Eau Galle Reservoir. From a comparison of annual ST and SC rates, sediment resuspension appears to be confined to shallow depths (i.e., ≤4 m) in this reservoir. In contrast, since annual ST and SC rates at depths >4 m were similar, resuspension and redeposition of profundal sediments could not be used as a likely explanation for increases in daily ST rates observed at depths >4 m during autumnal overturn.

Basin morphometry may be an important factor in resuspension of only the shallow sediments, as erosional energy becomes dissipated at deeper depths. Evaluation of wave dynamics in relation to basin morphometry suggests that wave scour, and sediment resuspension, is potentially greatest in shallow regions of lakes (Carper and Bachmann 1984). From differences in elemental ratios between surficial sediments and sediment collected in traps, Dillon, Evans, and Molot (1990) also suggested that only shallow sediments were being resuspended in Blue Chalk Lake. In other lakes, variations with depth both in the sedimentation rates (estimated from sediment cores) and in the physical and chemical composition of sediment have been attributed to resuspension and erosion of primarily shallow sediments (Håkanson 1977).

During autumnal overturn, daily ST rates increased linearly as a function of depth below a depth of 4 m. Similar increases in sediment trap rates with increasing depth have been reported in deep basins of other lakes during autumnal overturn (Andersen and Lastein 1981); however, the origin of this sediment (whether shallow, deep, or both) has often been difficult to identify. We suggest that these depth-related increases in daily ST rates at depths >4 m were the result of sediment focusing, and not resuspension and redeposition of profundal sediments. Resuspension of only shallow sediments, and homogeneous redistribution of particles throughout most of the water column, could explain the linear increases observed at depths >4 m. Mixing of shallow sediments throughout the water column may be accomplished by internal wave activity and/or convective circulation patterns that develop during cold nights.

Our results suggest that water mixing during autumnal overturn periods can result in substantial sediment resuspension and transport. These transport phenomena have important implications for the accumulation of contaminants and nutrients in the deep basin of lakes.

Acknowledgments

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Little Missouri River Environmental Enhancement Project

by
David R. Johnson[1]

Background

Narrows Dam is a multipurpose project located on the Little Missouri River 6 miles north of Murfreesboro, AR. It was authorized by the Flood Controi Act of 1941, as amended by the Flood Control Act of 1944, for flood control and hydroelectric power. The authority to maintain parks and recreation facilities was added in the Flood Control Act of 1962.

Construction of Narrows Dam was started in 1951 and completed in 1953. Soon after completion, the Arkansas Game and Fish Commission (AGFC) started receiving reports of poor fishing. In 1954, the AGFC attempted to restock the Little Missouri River (LMR) downstream of Narrows Dam with native warmwater fish. The attempt failed. The following year, the AGFC started stocking the LMR with trout. 'nitially, they attempted stocking yeai round, but 5,000 trout fingerlings stocked in July died within 1 week. The following year, 10,000 trout stocked in August died shortly after being released.

In 1961, the AGFC requested a minimum release of 1-hr of generation each day of the week in order to establish a year-round trout fishery. In 1962, an agreement was signed by the Southwest Power Association, the Corps, and the AGFC. The agreement called for minimum daily releases during July, August, and September to aid in the establishment of a year-round trout fishery. A 3-year trial program was established. Again, the AGFC stocked trout in the summer, only to have the previous results repeated. Thereafter, the AGFC has maintained a seasonal put-and-take trout fishery from October through April.

In 1970, the AGFC and the U.S. Fish and Wildlife Service (FWS) requested warmwater releases for the LMR. This request was repeated in 1975 and 1983 with an additional request for a minimum release. In 1972, the U.S. Congress held hearings on the Ouachita River basin in response to the request for warmer water releases from the dams on the Ouachita and Little Missouri Rivers. The outcome of the hearings was a Congressionally funded study on the Ouachita River basin.

Arkansas Lakes Interim Study

The Arkansas Lakes Interim Study was authorized in 1972 and covered the five lakes and three rivers in the upper Ouachita River basin. These included Lakes Ouachita, Hamilton, and Catherine on the Ouachita River; DeGray Lake on the Caddo River; and Lake Greeson on the Little Missouri River. Planning studies were initiated in 1976, and water quality monitoring was initiated in 1980. The water quality monitoring program consisted of installing and maintaining 18 in situ monitoring stations and obtaining weekly profiles in the five lakes

[1] U.S. Army Engineer District, Vicksburg; Vicksburg, MS.

of the study. The in situ monitoring program collected temperature, dissolved oxygen (DO), conductivity, and hydrogen ion concentration (pH) every half-hour for 3 years. In addition to the water quality monitoring, a fisheries study was conducted in the basin.

Results of the study showed that, in addition to low temperatures, the rivers downstream of the dams suffered from low DO concentrations and drastic diurnal flow fluctuations. The fishery study showed that the fisheries in the Ouachita River downstream of the three lakes and the Little Missouri River downstream of Narrows were severely impacted. In the 1975 request for warmer releases, the FWS had provided target temperatures and temperature ranges acceptable to the native warmwater fishes. The temperature of the releases from Narrows Dam and Blakely Mountain Dam fell below the required minimum monthly temperature by as much as 12 °C, and the temperature fluctuated as much as 15 °C daily.

In addition to low temperatures in the releases, the DO in the releases also fell below the minimum DO requirements of warmwater fishes for 2 to 3 months each year. Figure 1a shows the observed DO immediately downstream of Narrows Dam during the 3 years of the Arkansas Lakes Interim Study. The in situ data provided a much better insight into the problems of the system, but also illuminated the need for water quality modeling, which would require water chemistry data. Collection of these additional data began in 1983. The two data sets then provided the necessary data to run water quality models on Lake Greeson and the Little Missouri River. Lake Greeson was modeled with CE-QUAL-R1 and CE-THERM-R1 and the Little Missouri River was modeled with CE-RIV1. The water quality models were used to test a change on the outlet elevation of Narrows Dam. Figure 1b shows the DO in the releases from Lake Greeson with four outlet elevations, and Figure 2 shows the temperature in the releases with the same four outlet configurations. These studies showed that although any of the three elevated outlet configurations would provide significant temperature increases, only the highest two outlet elevations would provide adequate DO concentrations in the release waters.

Little Missouri River Environmental Enhancement Project

In order to be successful, the environmental enhancement project had to address the three major water quality problems in the LMR downstream of Narrows Dam. These problems were low temperatures, low DO, and drastic daily changes in flow. The temperature and DO problems were addressed by replacing the lower penstock trash racks with bulkheads. This would raise the elevation of the withdrawal zone and provide warmer release water. The water would be drawn from the epilimnion and metalimnion instead of the hypolimnion and metalimnion. Drawing water from the epilimnion would also raise the DO levels. Although the modeling results indicated that the DO would be substantially increased, the levels would still be below the U.S. Environmental Protection Agency minimum requirements. To further increase DO levels, the restoration plan calls for the random placement of boulders in the upper 0.5 mile of the stream. These boulders would create local areas of turbulence and enhance the reaeration of the river.

The final aspect of the restoration plan is to augment the minimum low flow. Under normal conditions the flow ranges from 3,000 cfs during hydropower generation to 15 cfs from turbine leakage during periods of nongeneration. The FWS and the AGFC considered 50 cfs to be a much better minimum flow. To increase this minimum flow, three low-head

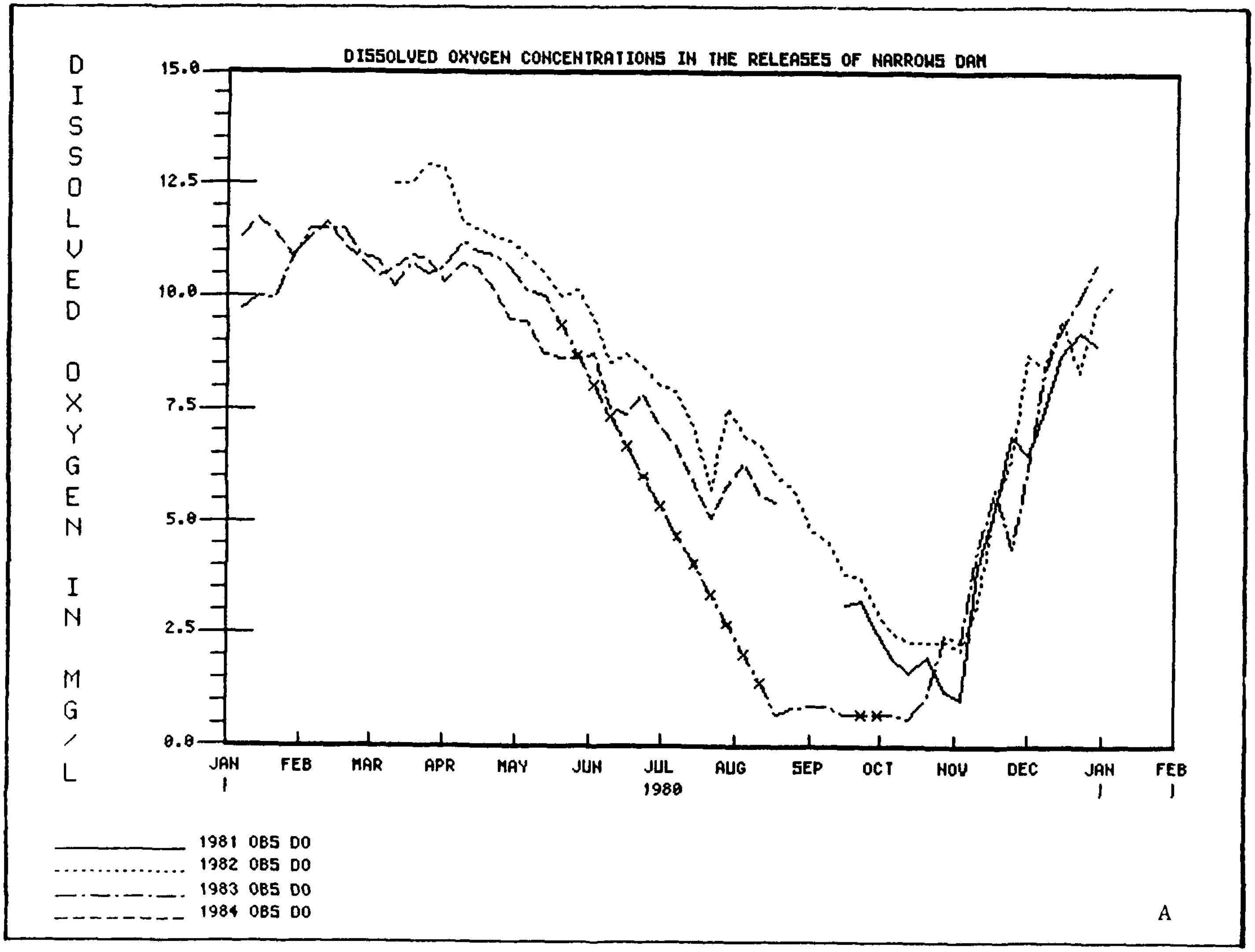

Figure 1. Measured and predicted dissolved oxygen concentrations in Little Missouri River (Continued)

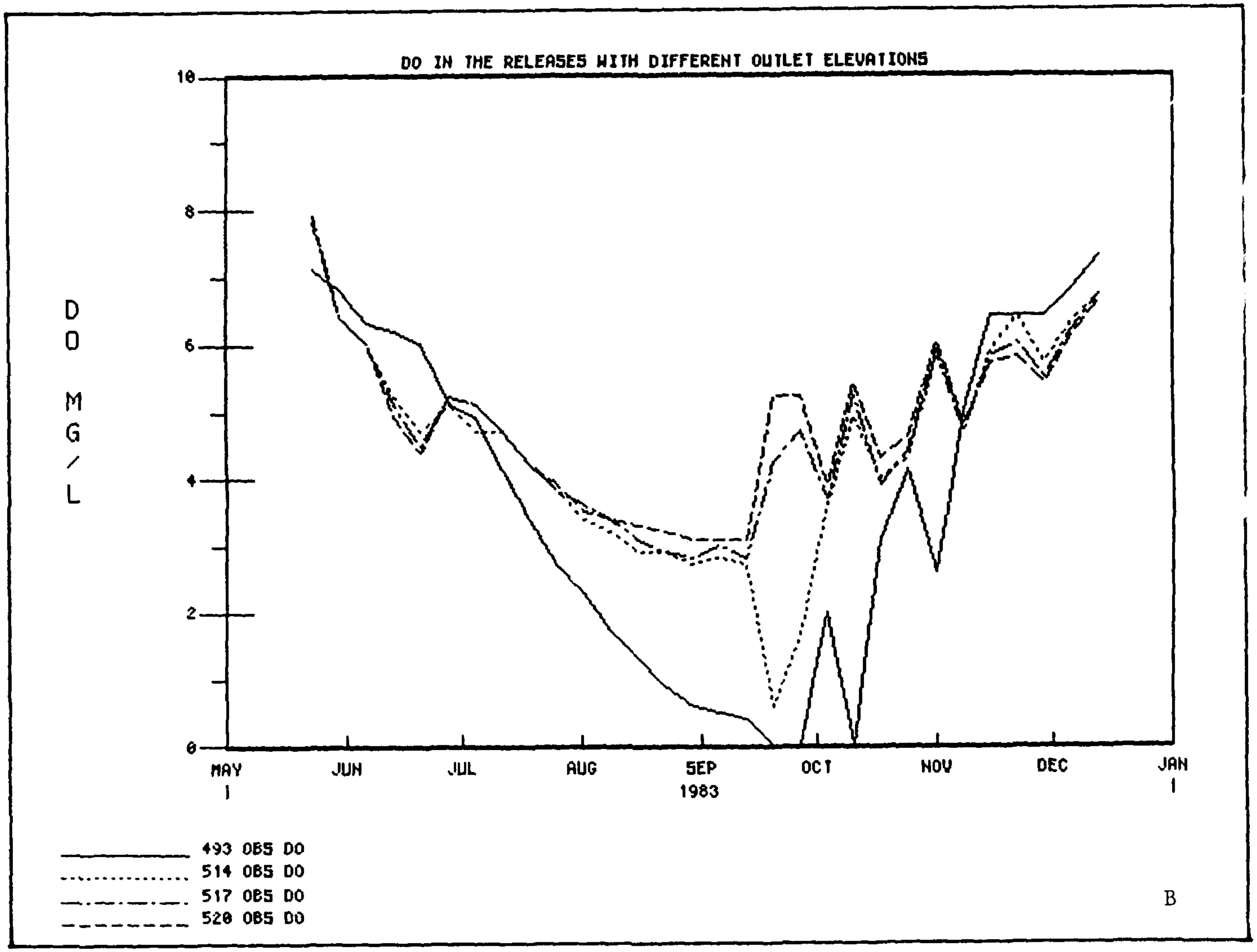

Figure 1. (Concluded)

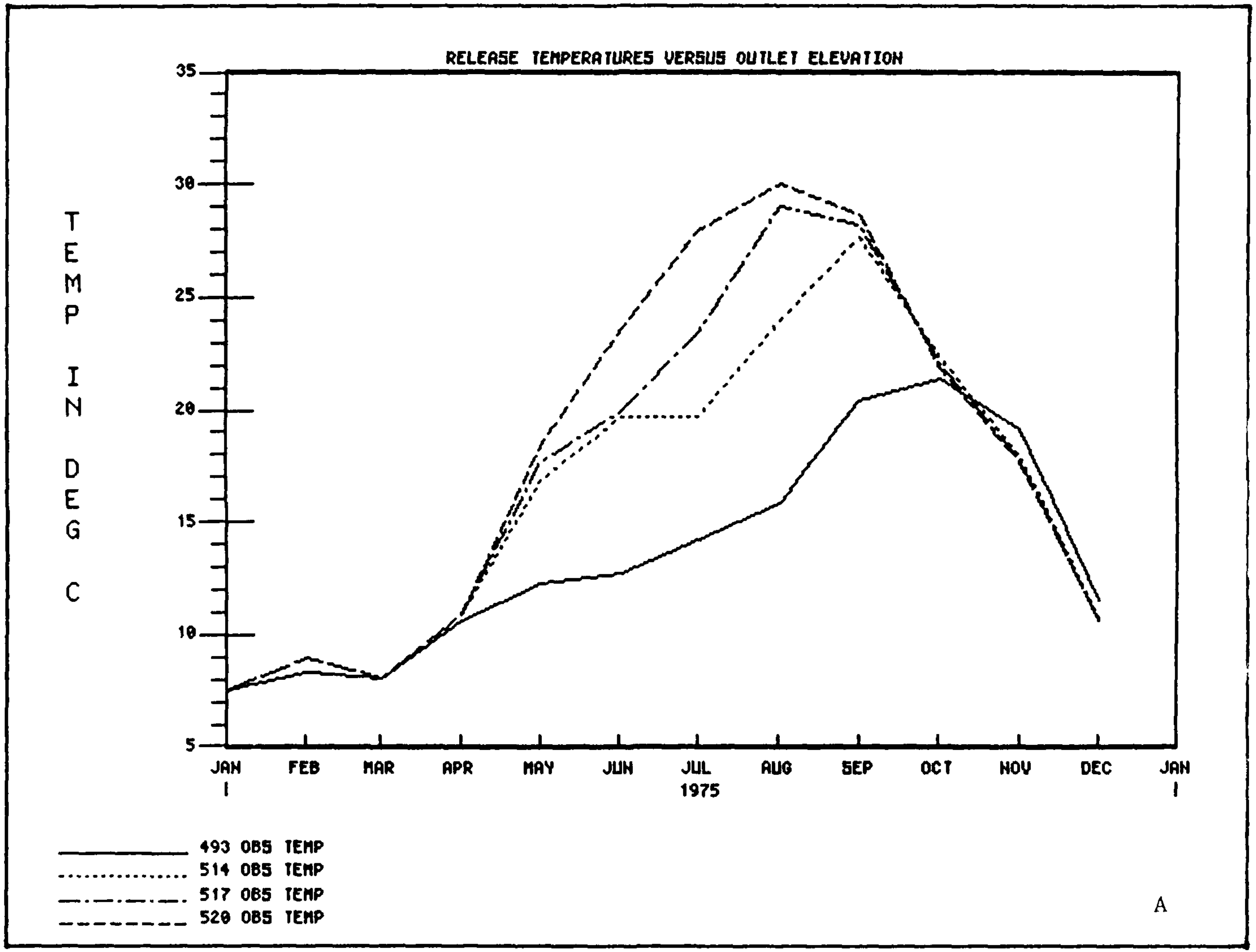

Figure 2. Predicted temperature versus outlet elevation (Continued)

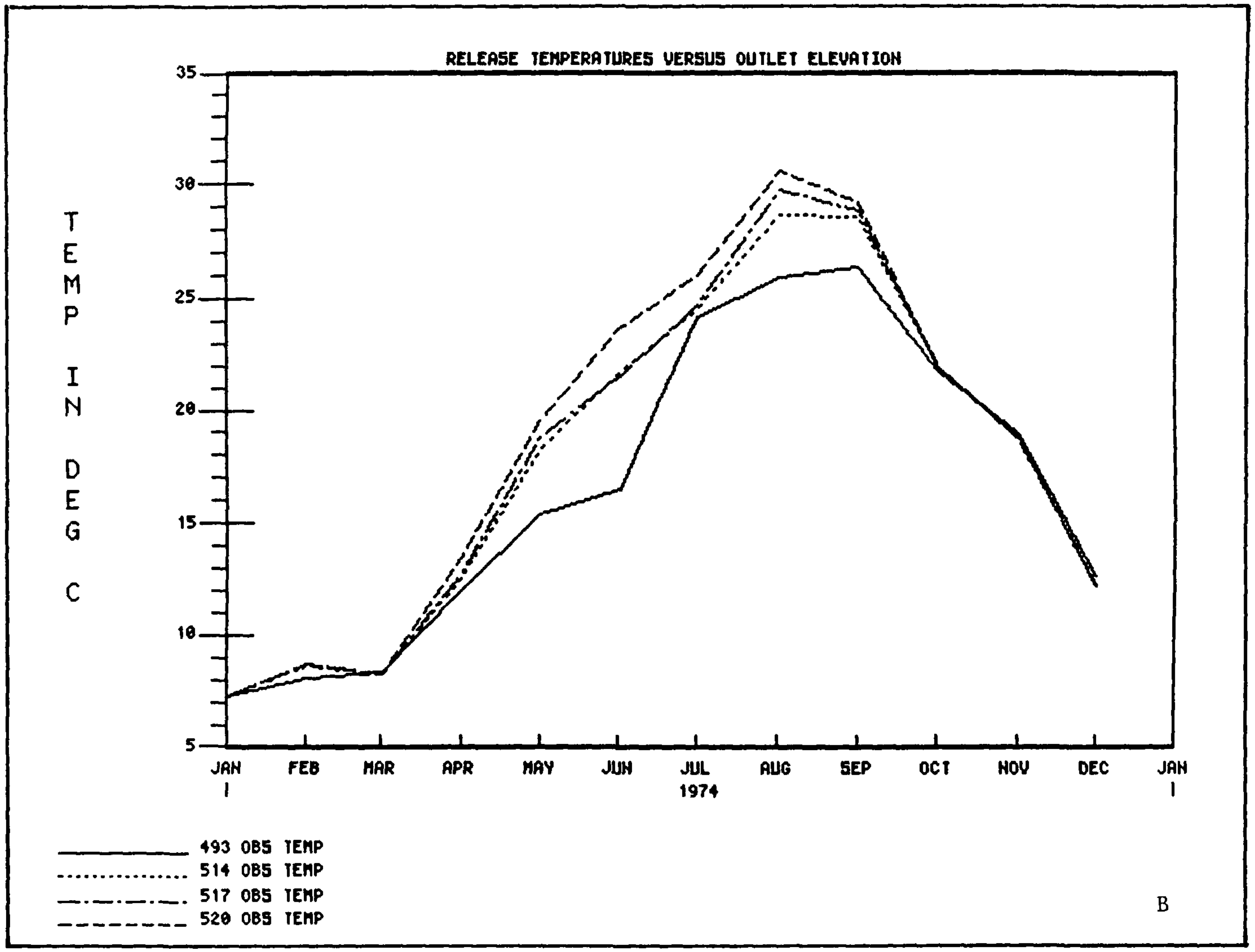

Figure 2. (Concluded)

rock weirs are to be constructed. The weirs will trap a small volume of water, releasing it gradually. Sufficient volume would be stored to maintain 35 cfs for 2 days. Thus, with normal leakage, the minimum flow would be increased to 50 cfs.

The Arkansas Lakes Interim Study ended by making recommendations for changes, but the District had no means to implement them. Congress provided a means with the Water Resources Act of 1986. Section 1135 of the act allowed the Government to spend up to $25 million to restore rivers and streams that had suffered environmental damage as a result of Federal projects. In January 1987, the Vicksburg District submitted the Little Missouri River restoration as a Section 1135 proposal, with the Arkansas Game and Fish Commission as the local sponsor. Pleased with their success, the AGFC wanted to announce their plans at a public meeting.

At the public meeting, held in Murfreesboro in April 1987, the AGFC's joy of achievement was dashed. A group of local trout fishermen expressed total opposition to the proposal. Instead of reestablishing the native warmwater fishery, the group wanted a year-round trout fishery. The AGFC took a step back and offered to complete a 1-year study to determine if a year-round trout fishery would be possible. The study supported the earlier studies. No trout were found in the LMR after May, and the temperature in the LMR was too high during periods of the day to support trout. While the study was being conducted, the trout fishermen waged an effective letter-writing campaign. They wrote to individual Commissioners, Congressmen, Senators, and the Governor of Arkansas. In their letters, they even proposed two alternate plans for ensuring a year-round trout fishery.

For 3 years no further actions transpired on the proposal, as no funding was provided until fiscal year 1991. It was then that the Vicksburg District learned that the 1135 proposal had been accepted and that complete plans were required with a signed Local Cooperation Agreement for cost sharing.

The District and the AGFC both were still strongly in favor of the project, but both parties were reluctant to move too rapidly. The District and the AGFC agreed that a second set of public meetings would be necessary. To avoid the mistakes of the first round, the two agencies carefully planned their course of action. The District and the AGFC jointly prepared a briefing. In January 1991 the briefing was first presented to the State Game and Fish Commissioners, when they approved the project. The briefing was taken to Congressmen and Senators, who were given the same briefing on a one-on-one basis. In this fashion, their support was obtained prior to any public announcements. Finally, three public meetings were scheduled. The first two were in downstream areas of the LMR, where the warmwater fishery had been most severely damaged. The final meeting in Murfreesboro was well attended. After a short presentation, the floor was opened to comments. The year-round trout fishermen were present, but they were outnumbered 5 to 1. Sentiment was overwhelmingly in favor of the project, with many asking that it be expedited. The final proposal has been returned to Washington, and the District is waiting for the final approval and the funds.

The most significant benefit from the work on the LMR will be the restoration of the native warmwater fishery in the Little Missouri River. A secondary benefit achieved is the good working relationship the Vicksburg District achieved as a working partner with the AGFC.

Applications of the Cumberland Basin Reservoir System Model for Water Quality Control

by
Jackson K. Brown[1]

Introduction

The Nashville District operates a system of eight major multipurpose dams in the Cumberland River basin in Tennessee and Kentucky. Experience has shown that regulation of these projects has a major impact on water quality conditions on the main stem of the Cumberland River. To be able to control water quality through project operations, it is necessary to have the capability to forecast the impacts of potential operations. This has led to the development of a reservoir system model for water quality control.

Description of Reservoir System

Four of the reservoirs with which the model is concerned (Wolf Creek, Dale Hollow, Center Hill, and J. Percy Priest) are classified as tributary storage projects and provide flood control benefits. The remaining four projects (Cordell Hull, Old Hickory, Cheatham, and Barkley), which are located on the main stem of the Cumberland River, are classified as run-of-river navigation impoundments. All eight projects produce hydropower and provide recreation benefits. A schematic drawing showing the relationships of the projects to each other is provided as Figure 1. Statistical information on the projects is provided in Tables 1 and 2.

Description of Reservoir System Model

The reservoir system model is not a single computer program, but actually consists of several relatively small programs and models. Some of the most important tasks performed by these programs and models include the following:

a. Simulate regulation of storage projects (through change in storage computations).

b. Route flows through main stem navigation projects (using coefficient method of routing).

c. Compute hydropower production in terms of peak, off-peak, and total energy, capacity, and monetary value.

d. Route temperature and dissolved oxygen (DO) through tailwaters of storage projects.

e. Mix inflows (water quality parameters are flow-weighted).

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[1] U.S. Army Engineer District, Nashville; Nashville, TN.

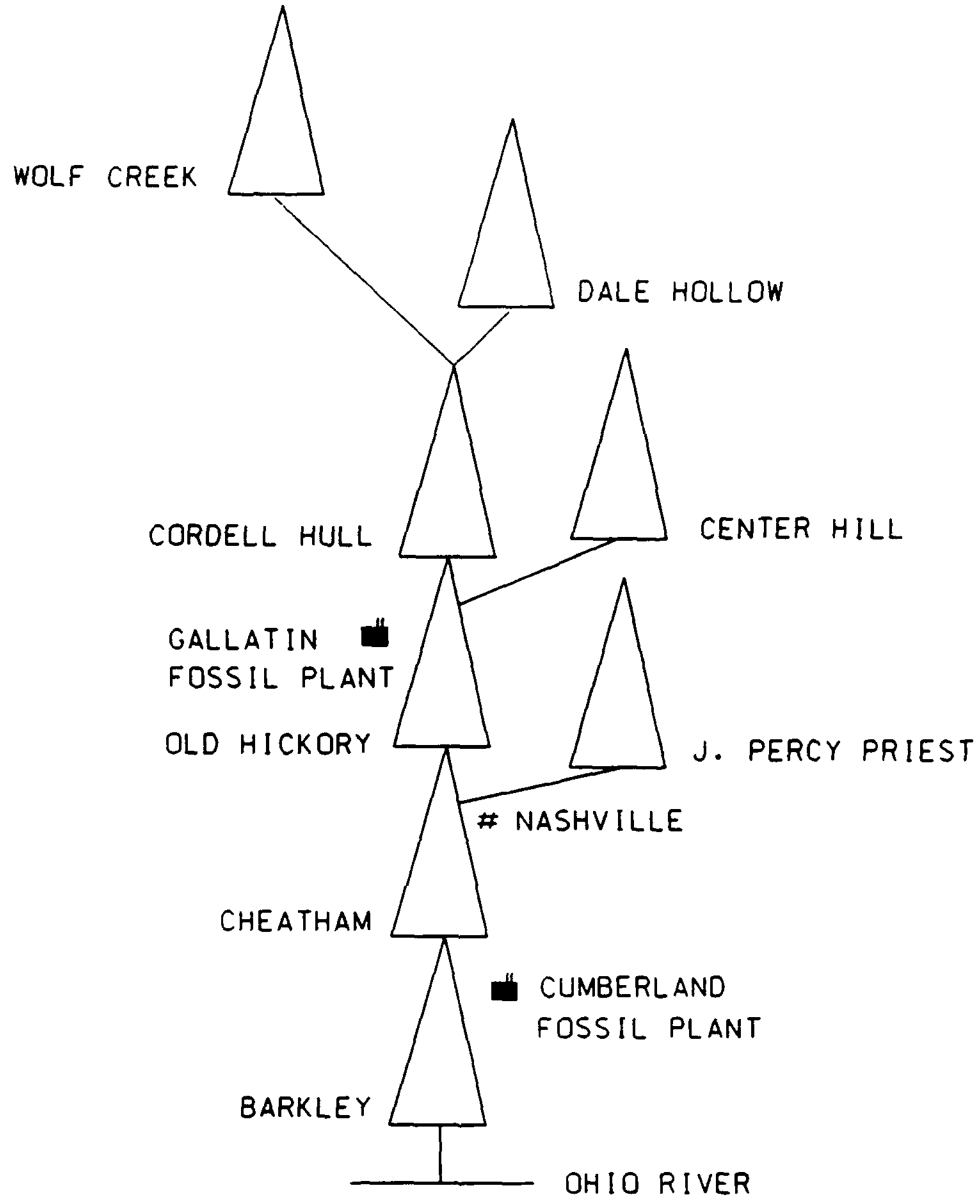

Figure 1. Cumberland Basin Reservoir System
Model--schematic configuration

f. Route temperature and DO through main stem navigation projects and compute outflow temperature and DO.

The first two tasks provide the capability to simulate the operation of the reservoir system from a hydraulic standpoint. In a system of hydropower projects without selective withdrawal, controlling flows is the only practical way to control water quality in real-time operations. Thus, flow is not only an important hydraulic parameter, but it is also an important water quality parameter as well. Hydropower (task c above) was added to the system model because of the need to evaluate impacts of proposed operations on energy production and capacity. Since hydropower was an authorized project purpose and water quality was not

Table 1
Storage Project Data

Project	Drainage Area sq miles	Surface Area acres	Volume acre-ft	Maximum Depth ft
Wolf Creek	5,800	50,000	4,000,000	185
Dale Hollow	900	28,000	1,400,000	155
Center Hill	2,200	18,000	1,300,000	175
J. Percy Priest	900	14,000	400,000	100

Table 2
Navigation Project Data

Project	Length of Backwater miles	Surface Area acres	Volume acre-ft	Maximum Depth ft	Avg Annual Discharge cfs
Cordell Hull	72	12,000	260,000	80	15,000
Old Hickory	97	22,000	420,000	70	19,000
Cheatham	67	7,000	100,000	40	23,000
Barkley	118	58,000	870,000	75	36,000

considered an authorized purpose, it was necessary to prove that operations for water quality control would not significantly impact hydropower.

Another important component of the model is the database of typical values for a number of hydrologic, meteorological, and water quality parameters. A significant effort has been make in analyzing historical data to define typical and, in some cases, extreme (maximum or minimum) values.

Applications for Real-Time Forecasts

The most important use of the model to date has been for real-time forecasts of water quality. Under normal operations a 10-day forecast is made weekly, using the flows resulting from the hydropower generation schedule. If water quality problems are anticipated, the model is used to develop a plan of operation to alleviate them. The model permits the volume

of water needed for various purposes to be quantified, which is preferable to just asking the water control managers for "more water." Experience has shown that proposed operations for water quality are more likely to be implemented by our water managers if impacts on other considerations such as flow, hydropower, and drawdown of the storage projects are presented with the water quality analysis.

Executing the model for real-time forecasts requires staying abreast of current conditions. The water quality monitoring program includes the collection of instantaneous temperature and DO data from the outflows of the four storage projects on a monthly basis. In addition, water quality monitors maintained through a contract with the U.S. Geological Survey record temperatures and DO concentrations at three sites on the main stem of the Cumberland. This information is transmitted to our office by GOES data collection platforms and is used to compute daily flow-weighted means. The only meteorological parameter required by the system model is air temperature. The National Weather Service provides the observed mean daily air temperature at Nashville and a 5-day forecast. Hydrologic data, consisting of flows and water surface elevations, are available from all of the dams in the system and a number of stream gages.

This information, as well as model input and output, is stored in the HEC DSS database. A considerable effort has been made to develop macros that permit information to be rapidly retrieved and displayed in a meaningful fashion. Some of the more useful plots include the following:

a. Historical median versus observed monthly average inflow--useful in assessing the severity of a drought and may be helpful in forecasting future inflows.

b. Typical versus observed and predicted daily headwater elevations--shows the availability of water in storage for providing supplemental flows and how proposed operations would affect the drawdown of the storage projects.

c. System weekly energy production versus minimum goal--shows whether a proposed operation will meet minimum hydropower production goals.

d. Observed instantaneous versus typical outflow temperatures and DO concentrations at storage projects--used to estimate past and to forecast future outflow conditions.

e. Observed versus computed outflow temperatures and DO concentrations at navigation projects--primary basis for adjusting model coefficients to reproduce actual conditions.

Applications During Drought of 1988

A major test of the model's predictive capabilities for water quality occurred in the drought of 1988. This was the first time that water quality was a major factor in determining system operations for the Cumberland basin reservoir system. The water quality parameter of most concern was DO. From previous analyses it was known that the critical point for DO in the system of main stem projects is the Old Hickory outflow. Although the primary objective was to release sufficient water from the storage projects to maintain the outflow DO concentration at Old Hickory above 5 mg/L, it soon became obvious that insufficient water was available in storage to permit this. Once this was determined, the objective was changed to maximizing the minimum outflow DO concentration. The model indicated that sufficient

water was available to prevent the mean outflow DO from dropping below 4 mg/L, and this became a primary operating objective for the system for the remainder of the summer. Data collected by the Old Hickory tailwater monitor indicate that operations were successful in achieving this goal. The mean outflow DO at Old Hickory dropped to near 4 mg/L on several days but did not drop below this level.

Intake Temperature Forecasts for Generating Plants

The model is also being used for real-time forecasts of intake water temperatures at two Tennessee Valley Authority (TVA) coal-fired generating plants. (The Gallatin fossil plant is located on Old Hickory Lake, and the Cumberland fossil plant is on Lake Barkley.) TVA's goal is to anticipate flow and water temperature problems that would cause the plants to violate their NPDES permits, so that operations can be scheduled to prevent such violations. This program was initiated on a trial basis in 1991 and will be continued in 1992.

Drought Contingency Plans

The model has also been found to be useful in the development of drought contingency plans. The three major operational considerations during droughts are (a) water available in storage, (b) water from future inflows, and (c) instream flow needs. The volume of water available in storage at a project is determined by the current headwater elevation and the knowledge of how much the project can be lowered in the future.

The volume of water that will be provided by future inflows is an unknown which will depend primarily on future climatological conditions. In view of the uncertainty of predicting future inflows, it is likely that several model runs will be made with assumed inflows ranging from typical to some worst-case scenario. As mentioned previously, the model computes observed inflows, and routines are available to compare them with typical values. Thus, it is possible to compute the ratio of observed inflows to typical inflows for a specified time period and express the result as a percentage of typical inflow. This is a good index for describing the severity of a drought and may be useful in developing an inflow scenario for one of the model runs in which the drought continues at the same level of severity.

The model has also been used to determine instream flow needs for hydropower and water quality. For hydropower, the objective was to determine the flows required to meet minimum weekly energy goals. It was assumed that all projects followed their typical headwater elevations and local inflows were normal. For water quality, the objective was to determine the flows which maintained the DO in the Old Hickory outflows at 5 mg/L. It was assumed that inflow quality was normal and meteorological conditions were typical. Results are shown in Figure 2. It is interesting to note that the patterns for these two purposes, which are often considered to conflict, are actually quite similar.

Future Development

Development of the system model is expected to continue. Work is currently being performed through the Waterways Experiment Station to apply optimization routines to the model. In its present form, the model will only simulate the results of a specified method of operation. Optimization will give the model the capability of developing its own regulation

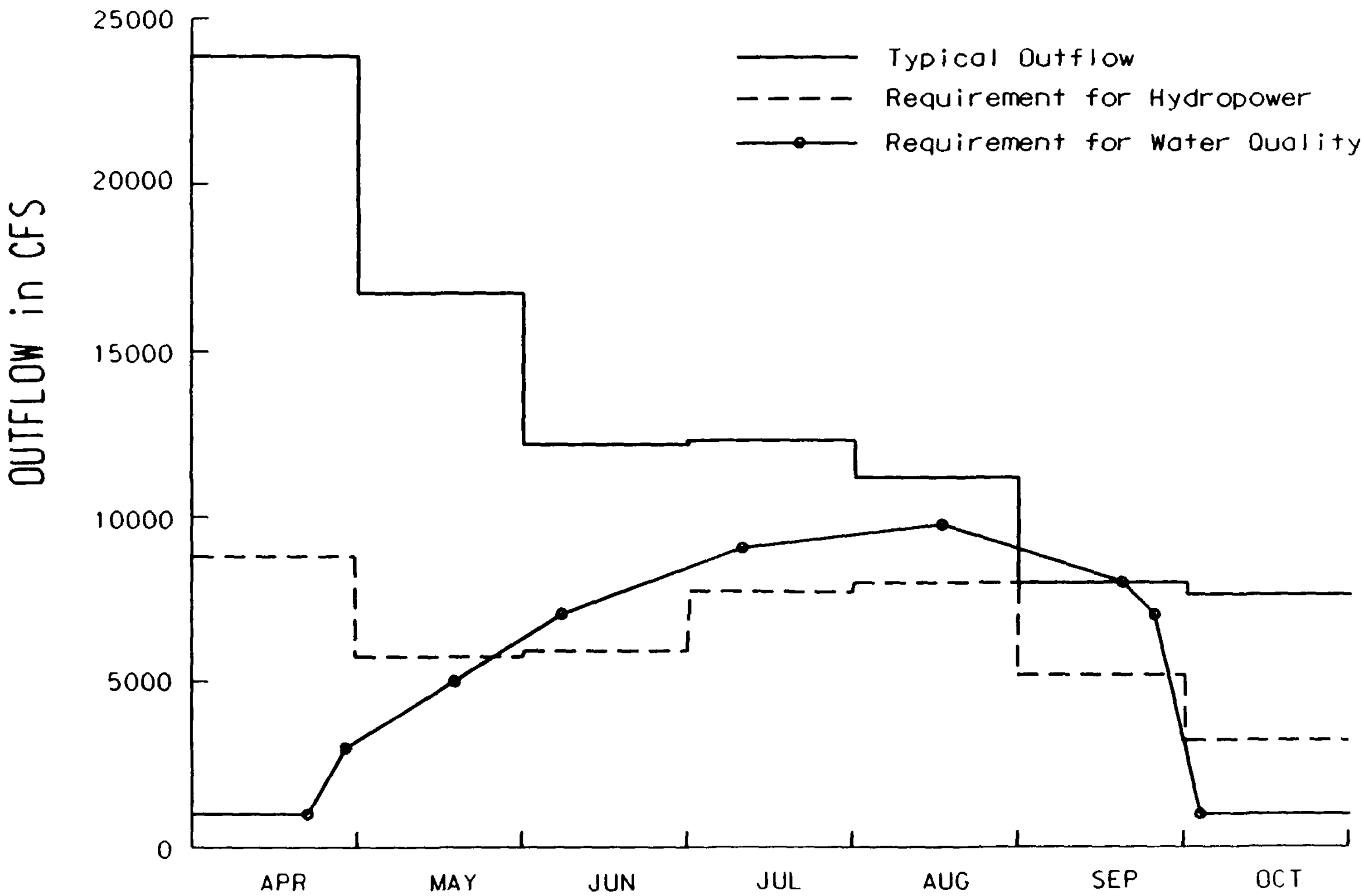

Figure 2. Old Hickory Dam--outflow requirements for hydropower and water quality

plan to meet operational objectives. Plans also call for development of the capabilities to simulate impacts of point sources of pollution on DO and to route spills through the four navigation projects.

Management Technique for Long-Term Flow Augmentation

by
Kenneth S. Lee[1]

Introduction

Flow augmentation is an important aspect of reservoir operations in times of low flow. The primary purposes of flow augmentation are to (a) protect the stream habitat below dams so that aquatic biota can be maintained during dry periods, (b) improve fish habitat, (c) improve stream quality, and (d) provide adequate water supply. Although most projects have established some form of augmentation flow, the method for determining augmentation flow differs from project to project.

In circumstances in which no storage is authorized for augmentation flow, minor pool fluctuation can often be used to provide some flow augmentation.

When a reservoir project is authorized storage for augmentation flow, heavy use of this storage will likely impact the other authorized project purposes. Generally, hydrological studies as well as water quality studies are conducted to determine efficient use of the storage for maximizing overall benefits.

At some projects, an instream habitat model has been used to assess flow augmentation requirements. The goal of this modeling is to estimate the effect of water resources management activities on the habitat of aquatic biota in a flowing-water system. Many factors are involved in evaluating how best to manage a project for maximum downstream productivity.

The Instream Flow Incremental Methodology and Physical Habitat Simulation systems are widely used for this purpose. The model results indicate the relationship between habitats and flow volumes in free-flowing conditions. The model provides an optimum flow, but it does not consider reservoir resources, nor can it indicate how to arrive at the volume for flow augmentation when other factors must be considered. Model results can be expected to show that high minimum flow maintains habitats. In many cases, however, reservoirs cannot provide the flow recommended in the model results.

The purpose of this paper is to demonstrate a management technique for flow augmentation that is applied to the Jennings Randolph Lake project. This technique requires predicting reservoir inflow and effective storage use with time. The concepts demonstrated here may be useful to those who manage other projects with similar purposes.

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[1] U.S. Army Engineer District, Baltimore; Baltimore, MD.

Case Study: Flow Augmentation at Jennings-Randolph Lake

Jennings Randolph Lake is located on the boundary between Garrett County, Maryland, and Mineral County, West Virginia, on the North Branch of the Potomac River (Figure 1). The drainage area of this project is 263 square miles. The Jennings Randolph Lake project was authorized by the Flood Control Act of October 1962. Its purposes are to provide water quality control in the North Branch of the Potomac River, industrial and municipal water supply for the Potomac River basin, flood control protection for the North Branch communities, and recreation associated with the lake and surrounding facilities. The project provides 36,200 acre-ft of storage for flood control, 41,000 acre-ft of municipal water supply storage for the Washington, DC, region, and 51,000 acre-ft of water quality storage.

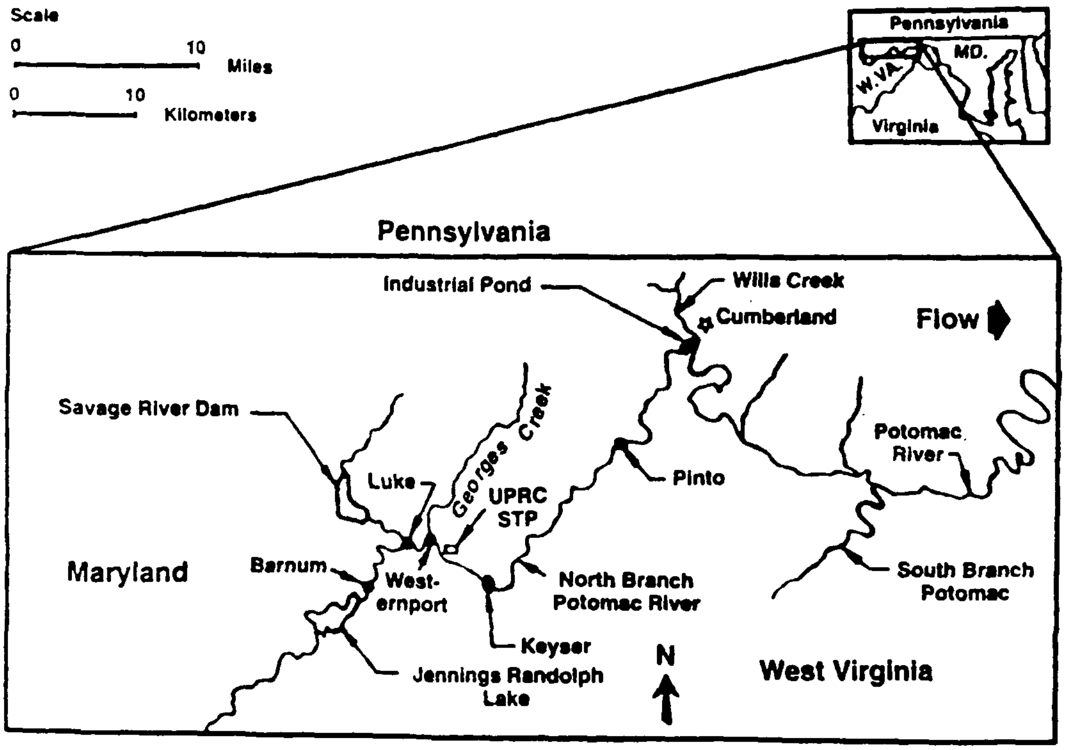


Figure 1. Study area map

The water quality benefits at Jennings Randolph Lake result from the dilution of acid mine drainage and industrial and municipal wastewaters and from the reduction of thermal pollution. Water quality control operations for these purposes often use 80 to 90 percent of the 51,000 acre-ft allocated to water quality storage each year.

Acid mine drainage both in the watershed and below the dam causes a water quality problem in the Jennings Randolph reservoir and downstream. Acid reduction in the reservoir is accomplished by storing inflow, to the extent possible, in the reservoir and by releasing carefully blended water through a selective withdrawal system that allows releases at six levels. Below the dam, acid reduction is accomplished by increasing outflow rate or discharging better water quality when high runoff occurs below the dam. Acid slugs can be expected with a pH as low as 4.5.

The North Branch Potomac River below the dam is also degraded by industrial and municipal wastes. A major source is a Westvaco paper mill, which is located 8 miles below the dam at Luke, MD. Its effluent contains high concentrations of dissolved and suspended solids with dark tanning color. The Jennings Randolph project is regulated to dilute the wastes by maintaining a flow as high as possible on a long-term basis.

The Westvaco paper mill produces heated wastewater both as a by-product of paper processing and from its coal-fired power plant. The heated wastewater requires a cooling process before it is discharged into the river. By regulating releases for low stream temperatures, the project is able to reduce the expense required for the cooling process at the Westvaco paper mill.

The water quality storage is used for supplementing flow augmentation during low-inflow periods and is refilled when inflow increases above a specified flow. Heavy use of the water quality storage frequently causes conflicts with other project purposes. Effective use of water quality storage is a key to achieving the best downstream water quality while maintaining in-lake quality and recreation.

Water control management at Jennings Randolph lake involves establishing priorities on the basis of project purposes. The priorities vary with the time of the year, hydrologic conditions, water supply needs, water quality conditions, and other factors to be considered in the operation of the project. On the basis of evaluations, a water control manager develops a plan to use the water quality storage with time.

Effective use of the water quality storage requires a real-time inflow prediction technique. This real-time inflow prediction is determined by a monthly flow frequency and a consecutive monthly flow frequency curve. The monthly flow frequencies are constructed by using monthly mean flows from 60 years of historical data at Kitzmiller, MD. The consecutive monthly flow frequencies are constructed using combined data from 2- to 5-month periods, depending upon the desired period of the prediction As an example, the consecutive flow frequency of July and August is constructed using the calculated mean of July and August inflow for each year for the period of record. Likewise, a consecutive flow frequency for a different time period is constructed by using the calculated mean inflow corresponding to that time frame. Figure 2 exhibits individual monthly flow frequencies, and Figure 3 exhibits consecutive monthly flow frequencies.

A current flow trend is derived from a monthly flow frequency by comparing recent flow data with historical flows from the same months. This flow trend is compared to other factors such as long-range meteorological forecasts at the area and is revised if needed. A future flow trend is derived by assuming that the inflow in coming months will follow the pattern of previous months. By applying a current flow trend to a consecutive monthly flow frequency, a future inflow is estimated.

When the future inflow is determined, storage management is greatly simplified. The reservoir is assigned operating ranges of pool elevation with time by considering project purposes together with the estimated inflow and an acceptable risk factor. Available storage for augmentation is calculated as the difference between storage available at the start of the period and the storage associated with a target reservoir elevation at the end of the period, and divided by the length of the period. The continuity equation is then used to determine an appropriate reservoir release, based on the estimated future inflow and the available storage.

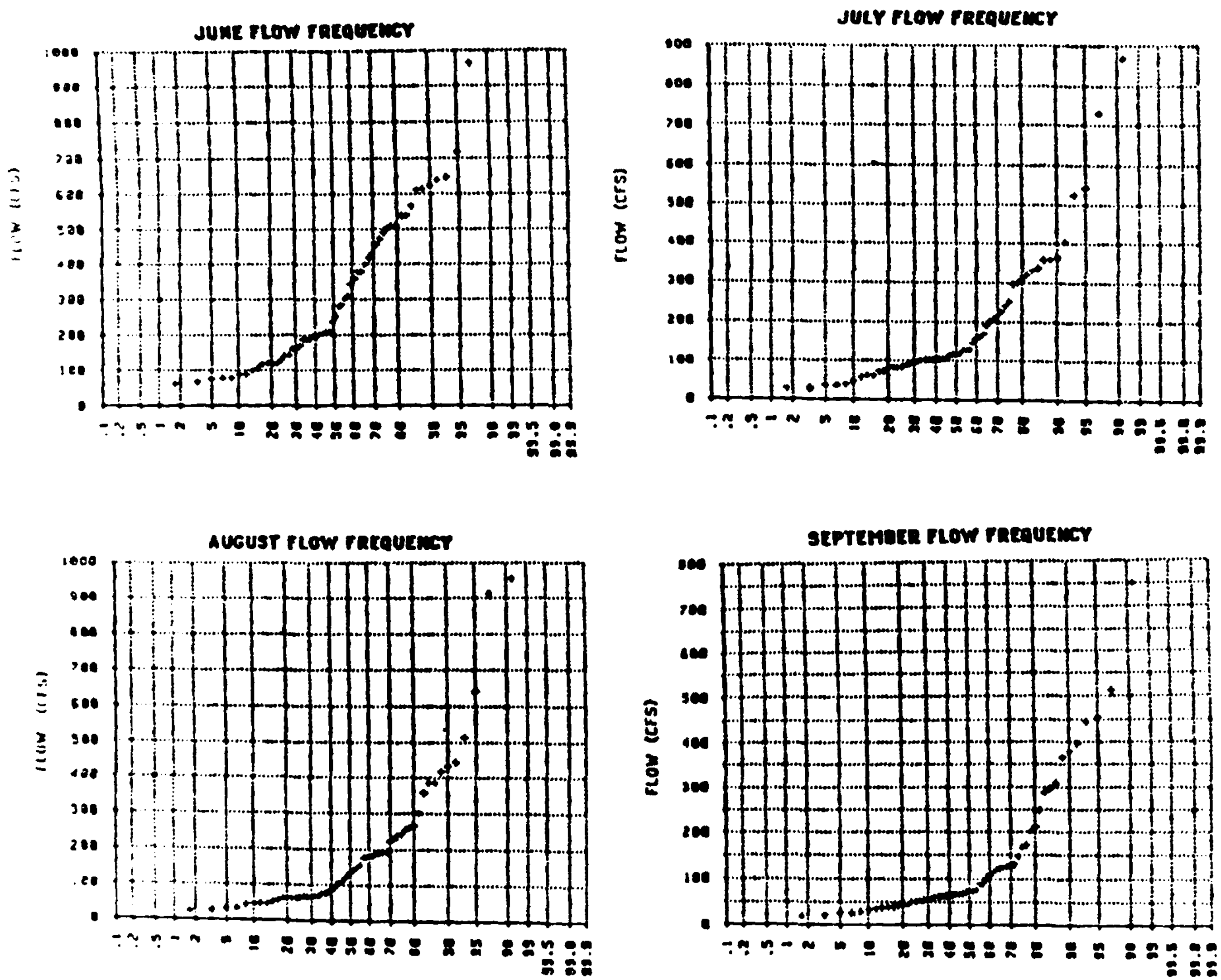


Figure 2. Monthly flow frequency (June-September)

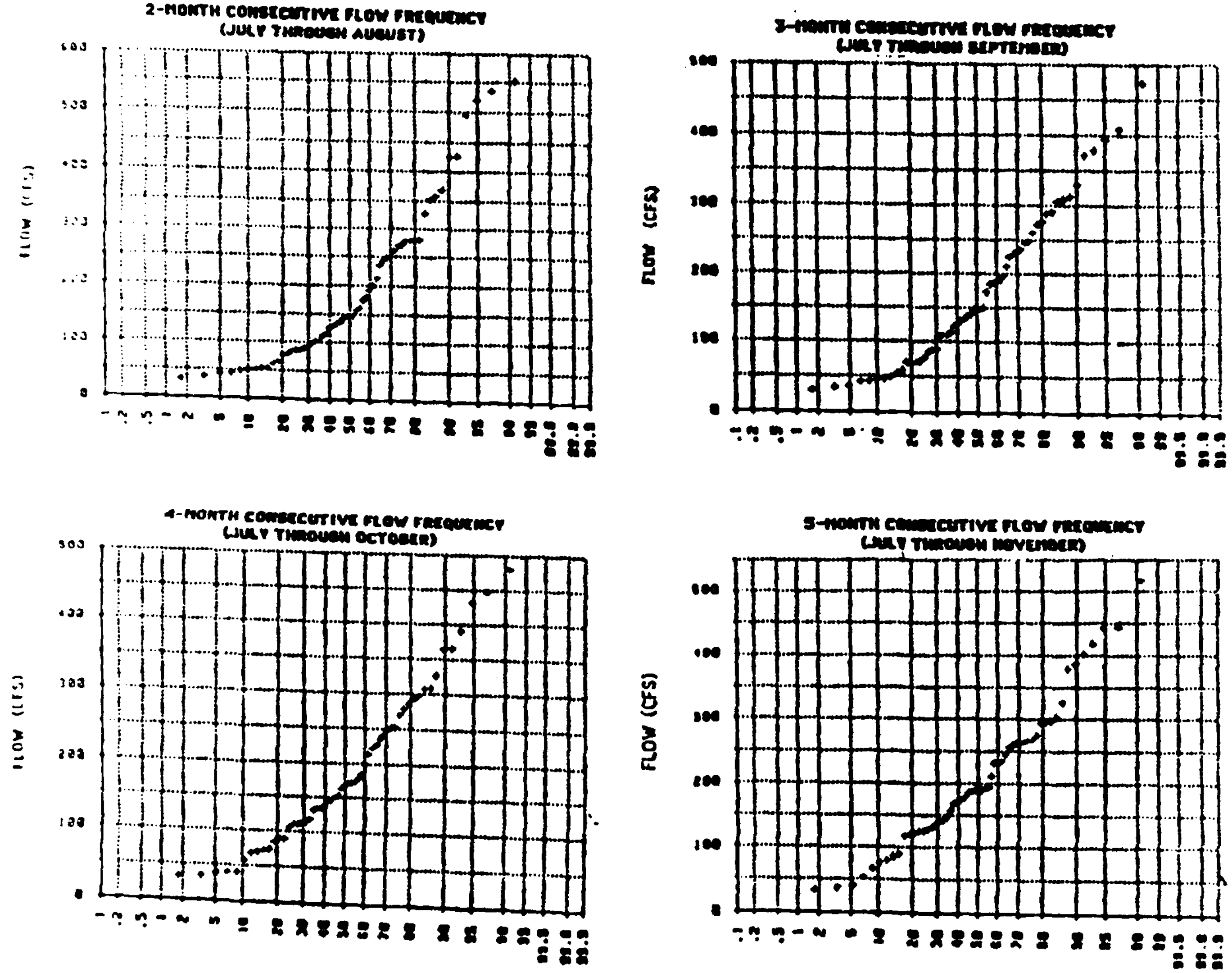


Figure 3. Consecutive monthly flow frequencies (2-, 3-, 4-, and 5-month periods)

Figure 4 presents an example of the operating ranges of pool elevation in Jennings Randolph Lake for water quality operations. Curve B exhibits the lowest pool elevation to meet a minimum outflow of 120 cfs, Curve A shows a maximum allowable pool for water quality control, and Curve C represents a guide curve for achieving overall project purposes.

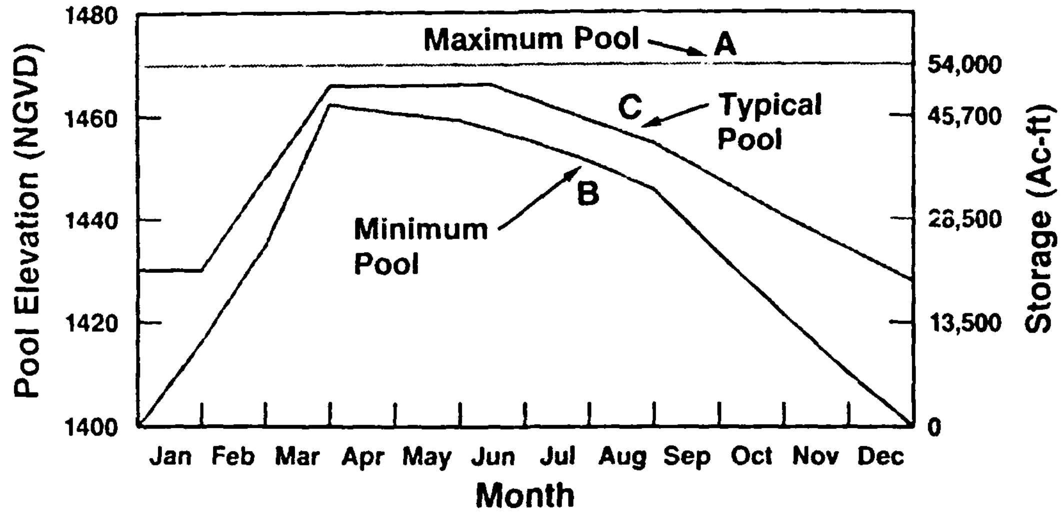


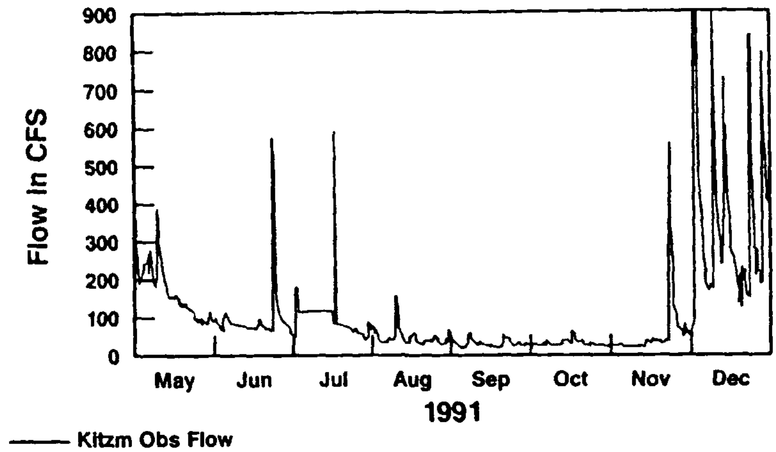
Figure 4. Operating ranges of pool elevation for water quality operations, Jennings Randolph Lake

Let's make an assumption. Today is July 1. The average June inflow was 110 cfs. June's inflow trend had a recurrence of 20 percent based on the historical data, using Figure 2. This result was compared to the long-range meteorological forecast of 30 days and 90 days from the National Weather Service, and the trend is adjusted if necessary. If we assume that the current inflow trend will continue for the next 3 months (July through September), the estimated future inflow would be obtained from a consecutive month flow frequency (Figure 3b) with a recurrence of 20 percent. From Curve C in Figure 4, the pool elevation is 1,460 ft NGVD (National Geodetic Vertical Datum) (storage, 89,100 acre-ft), and the target elevation at the end of 3-month period is 1,440 ft NGVD (storage, 71,800 acre-ft). The release rate would then be 164 cfs, which is 70 cfs from the estimated inflow and 94 cfs from the storage. The evaluation should be repeated every other week, and if the current inflow trend changes or a target elevation is changed, the release rate is adjusted.

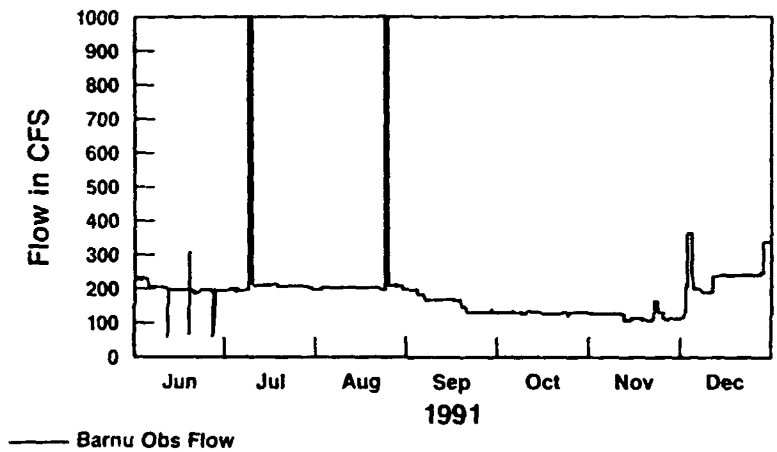
With this technique it is possible to provide high augmentation flow on a long-term basis with only minor impacts on other project purposes even during droughts. The Jennings Randolph Lake project has utilized about 80 to 90 percent of the water quality storage of 51,000 acre-ft every year. Figure 5 shows inflow, outflow, and pool elevation for 1991 at Jennings Randolph Lake.

Conclusion

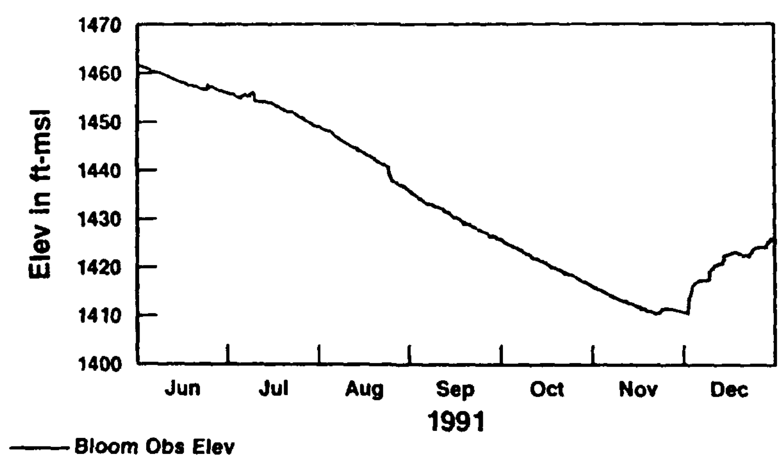
A simplified management technique to select reservoir releases for long-term flow augmentation has been developed by the Baltimore District. The technique provides for effective use of 51,000 acre-ft of water quality storage in Jennings Randolph Lake. Based on an evaluation of current flow conditions at the reservoir, an operating risk is assigned. This risk is



a. Observed inflow, Kitzmiller, MD



b. Observed outflow at Barnum



c. Observed elevation at Bloomington

Figure 5. Reservoir operations for 1991,
Jennings Randolph Lake

associated with mean inflow frequency curves developed from 60 years of historical inflow data for each possible combination of consecutive 1- to 5-month periods. Using the inflow frequency curve that is applicable for a given operating period, an estimated inflow for the target period can be estimated. An appropriate reservoir release is determined based on the estimated inflow and available storage. Generally, inflow conditions and available storage are assessed biweekly, and if warranted, the release is adjusted.

Implementation of Environmental Initiatives--from WRDA to Wetlands and Habitat Restoration Projects

by
Carol A. Coch[1] and Marshall G. Nelson[1]

Introduction

The U.S. Army Corps of Engineers' role as the Nation's "environmental engineer" is reflected by a shift in emphasis from more traditional construction projects to projects that are partly or wholly environmentally based. These environmental projects usually include a component of wetland or aquatic habitat restoration. Much of the authority for the Corps' environmental mission was derived from the Water Resources Development Acts (WRDA) of 1986 and 1990. Section 306, WRDA 1990, established environmental protection as one of the Corps' primary missions. Section 307, WRDA 1990, established the goal of "no net loss" of wetlands for Corps water resources projects. This paper describes the transition from Environmental Initiatives to funded wetlands, habitat, and environmental restoration projects in the North Atlantic Division. The Corps has clearly shown an interest and commitment to providing environmental benefits associated with Corps projects in the past such as beneficial uses of dredged material; however, new laws and regulations allow the Corps to focus on Environmental Initiatives in a variety of ways and with additional funding mechanisms. An excellent overview of these laws and regulations was given by Dave Mathis, Biologist, CECW-PO, at the Long-Term Management Strategy Forum (Mathis 1992).

Environmental Initiatives have been established in response to Federal environmental laws and regulations and the President's "no net loss of wetlands" goal. They offer a unique opportunity for environmental protection and restoration on both Military and Civil Works projects. Many of these Environmental Initiatives were authorized under WRDAs 1986 and 1990, and some have been funded. Other guidance discussed with respect to environmental projects includes Corps policy, such as the FY92 and FY93 Civil Works Budget Guidance, the draft Civil Works Hazardous and Toxic Waste Guidance, and Corps programs such as the Defense Environmental Restoration Program on Formerly Used Defense Sites (DERP/FUDS). The Corps' budget guidance allows for environmental restoration as a primary project purpose equal in importance to other traditional project purposes. This is a departure from previous policy. It does not guarantee that a project will be funded and carried out. The DERP/FUDS program allows for environmental assessment and cleanup of hazardous, toxic, and radioactive wastes (HTRW) on formerly used defense sites. The Corps also acts as the U.S. Environmental Protection Agency's (USEPA) contractor for the Superfund program.

The Environmental Initiatives have also led to cooperation with other Federal and State agencies to jointly achieve environmental restoration goals. Interagency programs in which the Corps is a partner or cooperating agency include the National Estuary Program, the Chesapeake Bay Program, the Coastal America Partnership, the U.S. Fish and Wildlife

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Service's (USFWS) North American Waterfowl Management Plan, the Memorandum of Agreement between the Corps and NOAA on fisheries habitat restoration,[1] and continuing interagency National Environmental Policy Act (NEPA) coordination on Corps projects. These interagency programs of both national and international scope are discussed further below.

North Atlantic Division Environmental Initiatives

Several case examples of North Atlantic Division (CENAD) projects that have been funded under the authorities of WRDAs 1986 and 1990, General Investigations, and the Operations and Maintenance (O&M) program will also be described in detail.

Chester River (Bodkin Island), which includes habitat restoration, wetlands creation, and island building, was approved in FY91 under Section 1135, WRDA 1986. Bodkin Island, Cowanesque Lake (Pennsylvania), and Aberdeen Proving Grounds (Maryland) involve monitoring and stewardship of wetlands and have been funded as demonstration projects under the U.S. Army Engineer Waterways Experiment Station (WES) Wetlands Research Program under Section 307 of WRDA 1990.

New start FY92 General Investigation studies include Susquehanna River and Lehigh River, Pennsylvania, fish habitat studies. These were authorized by Congress and included in the FY92 Budget in accordance with Engineer Circular (EC) 11-2-157. One example of environmental initiatives studies being funded as part of a Construction General project is the Norfolk Harbors and Channels Inner and Outer Harbor Long-Term Management Study (LTMS). The Norfolk Harbor LTMS includes beneficial uses of dredged material for wetlands and upland habitat creation in Chesapeake Bay. In addition, a feasibility study for environmental restoration authorized as a general investigation with some independent supplements under Section 1135, WRDA 1986, is being performed by the Baltimore District (CENAB) for the Anacostia River and Tributaries Project, Washington, DC (USAED, Baltimore 1991).

Other environmentally based programs that CENAD coordinates include CADD for Military Environmental Projects; Zebra Mussel research proposals; Corps support for the USEPA Superfund Program; the Aquatic Plant Control Program; the Legacy Resource Management Program for environmental and cultural resource management on military lands, in which CENAD supports the military as a contractor; CENAD review of Base Realignment and Closure NEPA documentation; CENAD review and direction of a feasibility study by the New York District on hydro-environmental monitoring and modeling of the New York Bight (database, monitoring, modeling) authorized under Section 728, WRDA 1986; the Dredged Material Placement Alternatives Study for the New York/New Jersey Harbor area authorized under Section 412, WRDA 1990 (includes Section 211, WRDA 1986, closeout report on alternatives and demonstration projects); and the Partners for Environmental Program Market Feasibility Studies for regional approaches to solid waste management linked to opportunities for privatization and the military Environmental Compliance Assessment System in which the Corps (CENAB for CENAD military) is responsible for external environmental assessments

[1] Cooperative Agreement Between the National Oceanic and Atmospheric Administration and the Department of the Army for a Program to Restore and Create Fish Habitat (1991).

on all active Army installations with goals of environmental compliance, restoration, and production in pollution.

North Atlantic Division
Case Examples

DERP/FUDS Program: CENAD is a review center for the Defense Environmental Restoration Program on Formerly Used Defense Sites. Together with the Districts, the Baltimore HTRW Technical Center, and other Corps technical centers, CENAD has implemented cleanup on sites throughout the Northeast. This includes unexploded ordnance, hazardous toxic and radiological wastes, containerized HTRW, remediation of leaking underground storage tanks (LUST), landfill, groundwater, and soil contamination cleanup on sites that were formerly used by the Department of Defense. These lands have been excessed and are slated or have already been turned over to state or local interests and private industry for use in schools, parkland, warehouses, factories, or the like. Some of these areas, such as Raritan Arsenal, New Jersey, were found to have HTRW contamination that is being cleaned up under DERP/FUDS.

After establishing that a site was actually owned and used by the Department of Defense, a preliminary site investigation is performed together with a record search to indicate whether it is likely that HTRW is present on the site. Recommendation is made for a site investigation if HTRW is suspected or known to be on the site. This includes testing of materials and chemicals in underground storage tanks and associated soil and other tests as appropriate. The results of the tests are analyzed, and recommendation for remedial action is made. Once approved, the appropriate District/technical center performs or contracts out the remedial action.

For example, live 500-lb TNT bombs were found popping up offshore at Assateague Island National Seashore. Those were removed immediately and detonated. Further investigation of the area showed that it had formerly been used by the Navy in World War II as a bombing practice range. When the range was closed down, excess bombs were buried in rows perpendicular to the shoreline. Although it was unlikely that anyone would dig down to the area where the bombs were buried, erosion of the shoreline over the past 50 years caused some of the bombs to be exposed. The recommendation was made that the remaining bombs be located, removed, and detonated.

Interagency Programs

National Estuary Program: The objective of this program, which is administered by the USEPA, is to clean up the Nation's polluted estuaries. CENAD is a cooperating agency on several National Estuary Program projects. This includes the Delaware Bay Estuary, the Hudson River/Raritan Bay Estuary System, and Chesapeake Bay.

The Coastal America Partnership (CAP): The CAP encompasses both coastal environmental restoration and source reduction of contaminants. In FY91, a proposal for decreasing phosphate-induced eutrophication in Long Island Sound was funded by USEPA. While the Corps has no direct participation in the initial projects in the CAP, a number of proposals have been discussed which could include the Corps in the future. It is hoped that both Baltimore and Norfolk Districts can participate in oyster reef and submersed aquatic vegetation restoration under this program in the future.

Corps/USEPA Chesapeake Bay Program: The Baltimore District Engineer and his staff participate in the committee structure that guides the Chesapeake Bay Program. This program involves the Governors of the bordering States and the pertinent Federal agencies in a program to clean up and improve the Bay environment. A direct involvement of the Corps has been the development with USEPA of a three-dimensional hydrodynamic and water quality model of the Bay, which is used to determine the effects of changing inputs to the Bay on water quality characteristics.

USFWS North American Waterfowl Management Plan (NAWMP): The NAWMP is an international agreement among the United States of America, Canada, and Mexico, the goal of which is to preserve and restore waterfowl habitat associated with migratory flyways. CENAD is a board member of the Atlantic Coast Joint Venture of the NAWMP. The Atlantic Coast Joint Venture includes coastal states from Maine to South Carolina. CENAD has assisted the USFWS in evaluating proposals for wetlands and habitat restoration and in implementing restoration on existing Corps projects that meet the goals of the NAWMP. This led to funding of the North Landing River, Virginia, proposal in FY91. Among the projects approved for FY92 funding by the North American Wetlands Conservation Council are North Landing River and Whitehurst Marsh, Virginia.

One of the requirements for funding of these proposals is completion of NEPA documentation. The emphasis for wetlands protection is for more than just ducks or seasonal waterfowl. Some of the criteria and ranking factors used for selecting projects for funding under the North American Wetlands Conservation Fund are intended to ensure that the broadest possible wetlands benefits are attained. Implementation of the NAWMP in the Atlantic Coast Joint Venture also includes both ongoing and proposed Military work. At Fort Drum, New York, $150,000 has been provided to USFWS to implement NAWMP activities on the 110,000-acre post. Aberdeen Proving Grounds, Maryland, has provided funds to manage wetlands. The APG goal is to establish more than 100 acres of wetlands through water control structures and drainage tile plugs.

The Corps is committed to implement the goals of the NAWMP on its existing Civil Works projects. Completed projects include Raystown Lake, Pennsylvania (Area 420), a 10-acre former farm crop area that was restored to wetlands by the Pennsylvania Game Commission in cooperation with USFWS and the Corps for a propagation area for mallard ducks and blue and green herons.

NOAA/Corps Cooperative Agreement

A Cooperative Agreement between the National Oceanic and Atmospheric Administration and the Department of the Army for a Program to Restore and Create Fish Habitat was signed in January 1991. This agreement resulted from a pilot study conducted by the National Marine Fisheries Service (NMFS) and the Corps which examined the practicability of establishing a nationwide habitat restoration and creation program using existing authorities, funding, and resources. Draft Corps Guidance on implementation of the cooperative agreement was promulgated in March 1991. The agreement combines NMFS and Corps policies on aquatic habitat restoration. The pilot study resulted in an oyster habitat creation project using O&M dredged material provided by Baltimore District in Chesapeake Bay.

Projects include habitat restoration and creation and conservation of coastal and anadromous fish habitats and wildlife habitat. Corps policy allows creation and restoration of fish

habitat, wetlands, and seagrass beds at existing projects when it can be accomplished properly without added cost and for those projects that can be cost-shared with non-Federal interests.

Section 1135, WRDA 1986

Chester River, Section 1135, WRDA 1986, FY91: Chester River (Bodkin Island) was funded for habitat creation in FY91. This is one of the project modifications for the improvement of the environment which is cost-shared 25 percent with the local sponsor. WRDA 1990, Section 304, authorized continuation of the 1135 program. Environmental restoration studies of Bodkin Island are also being performed as part of the Wetlands Research Program (Section 301, WRDA 1990).

Bodkin Island was 5 acres in size in 1954 and supported 106 black duck nests. It eroded to 0.9 acre of wetland and forest by 1986 and supported 54 black duck and mallard nests. Erosion eliminated most of the trees and the rookery for black duck since then. Since the 1950s, black duck populations have declined drastically as the result of habitat loss and other factors. The Corps, in conjunction with the State of Maryland Department of Natural Resources and the USFWS, will use dredged material from the Chester River project to create approximately 5 acres of wetlands at Bodkin Island and reestablish the black duck rookery. This will consist of 45,000 to 60,000 cu yd of dredged material, which will be placed, planted, and protected with stone to create habitat. The dredged material will be sculpted to create ponds, intertidal habitat, upland and tidal creeks, and plantings of marsh grasses and shrubs will be made to provide black duck nesting and brooding areas.

Section 307, WRDA 1990

Section 307d, WRDA 1990, directed that a 3-year Wetlands Research Program under the auspices of WES be established. Three CENAD proposal were funded: Chester River (Bodkin Island), described above; Cowanesque Lake environmental restoration; and Aberdeen Proving Grounds, which is a case example of stewardship of wetlands on U.S. Military property.

General Investigations

The Susquehanna River Fisheries Habitat Restoration Study, a FY92 New Start General Investigation that met the criteria for the FY92 Budget EC, was approved for the reconnaissance phase of study.

The Lehigh River Fisheries Habitat Restoration Study was approved for the reconnaissance phase of the study, a FY92 New Start General Investigation that met the criteria for the FY92 Budget EC.

The Anacostia Rivers and Tributaries Project is funded under General Investigations and Section 1135. The reconnaissance study was completed in December 1990, and the feasibility study, the purpose of which is predominantly environmental restoration, will be completed in 1994. The feasibility study will determine the best methods for restoring fish and wildlife habitat in the 170-square mile Anacostia River basin by the year 2000.

Major goals of the Anacostia River and Tributaries Project include removal of barriers to fish passage, tidal and nontidal wetland creation, planting of trees along the streambanks, and

cleanup of the river. This project represents one of the first Corps General Investigations studies which has environmental restoration as the major component and involves an entire river basin. Some of the major issues that will need to be addressed, on this and other Environmental Initiative projects, is the way in which environmental benefits will be determined and how the recommended alternative will be chosen and supported.

The Passaic River Main Stem Project, which is a flood control project in the Passaic River Basin, contains a significant wetlands preservation/restoration component. To maintain flood storage, more than 5,000 acres of wetlands will be preserved as part of the project. In addition, wetlands mitigation banking by the state of New Jersey has been proposed.

O&M-Funded Studies

The Long-Term Management Strategy for the Disposal of Dredged Material (LTMS) is a Corps policy for planning of O&M projects on the basis of a 50-year project life. One of the key components for many of the LTMS projects is environmental restoration.

CG-Funded Studies

An example of a LTMS study, which is funded under Construction General funds, is the Norfolk Harbors and Channels Inner and Outer Harbor Long-Term Management Study.

Conclusions

Many of the interagency and partnership arrangements have enabled the Corps to get "more bang for the buck," and most have resulted in an unprecedented level of interagency cooperation in achieving environmental restoration.

Two of the General Investigations studies proposed and approved in FY92 were Environmental Initiatives. Although it was somewhat disappointing to have over 40 proposals in this area and end up with two, the General Investigations studies approved in FY92 represent a major step in a new direction for CENAD. In some cases, we and our interagency counterparts have experienced some degree of frustration at the pace of approval for Corps environmental initiatives. However, the realities of funding, rather than a lack of desire to implement environmentally based projects by the Corps, have been recognized by the other agencies.

The Environmental Initiatives are enabling the Corps to expand its beneficial uses of dredged material program, which was constrained because of a lack of authority to allocate additional funds for environmental restoration if it was beyond the funding required for the primary project purpose. Participation, proposal criteria, size, completion time, and number of partners/interests are all factors in funding success to varying degrees, depending upon the program. With extremely tight budgets, the Environmental Initiatives that were funded first had the common characteristics of meeting the proposal/budget criteria, being generally small and doable within a reasonable time frame, and having specific environmental restoration goals, interagency support, and Congressional and/or other interest. We expect that as more of these environmental projects are successfully completed and their long-term viability is established, larger and longer term environmental restoration projects and programs will be approved. This will also improve our credibility with other Federal and state agencies that are involved in environmental resource management, as well as with the public.

Future Environmental Initiatives

Since the Corps' mission has expanded to include Environmental Initiatives, future work is likely to include Environmental Initiatives on Corps projects. It is likely that additional Environmental Initiatives will be implemented as policy guidance is developed for authorities that were given in WRDA 1990 but still require further guidance for implementation. For example, cleanup dredging as authorized in WRDA 1990, Section 312, in areas adjacent to but not part of Federal dredging projects, may become part of the Corps' program.

Construction of these Environmental Initiatives has enabled the Corps to become a respected leader in wetlands, habitat, and environmental restoration and protection. We intend to move forward on implementation of Environmental Initiatives in the future. Our future work is likely to include an even greater emphasis on Environmental Initiatives. We are encouraged with the progress so far on Environmental Initiatives, and on the execution of these valuable habitat restoration projects. Much remains to be done, particularly in the CENAD area, which has experienced severe ecosystem degradation in many areas as a result of the industrialization of our country. Many challenges await us, but we are determined to meet them and to continue to earn our place as the Nation's preeminent environmental engineers as we continue to fund and implement environmental restoration on both Military and Civil Works projects.

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Hazard Assessment and Modeling Activities for the ARCS Program

by
David C. Cowgill[1]

Abstract

The 1987 amendments to the Clean Water Act, in Section 118(c)(3), authorize the U.S. Environmental Protection Agency's Great Lakes National Program Office to conduct a 5-year study and demonstration project relating to the control and removal of toxic pollutants in the Great Lakes, with emphasis on the removal of toxic pollutants from bottom sediments. Human health, aquatic life, and wildlife hazard assessments are being performed for the Assessment and Remediation of Contaminated Sediments Program, focusing on the contributions to environmental hazards attributable to contaminated sediments. Contaminant mass balance modeling is being performed on the Buffalo (New York) and Saginaw (Michigan) Rivers. Summaries of these activities were presented.

[1] U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL.

Toxicity and Chemistry Testing of Great Lakes Sediments for the ARCS Program

by
Richard Fox[1]

Abstract

The 1987 amendments to the Clean Water Act, in Section 118(c)(3), authorize the U.S. Environmental Protection Agency's Great Lakes National Program Office to conduct a 5-year study and demonstration project relating to the control and removal of toxic pollutants in the Great Lakes, with emphasis on removal of toxic pollutants from bottom sediments. Comparative chemical, physical, and biological testing of freshwater sediment was performed. Biological testing focused on benthic community structure, acute toxicity, chronic toxicity, mutagenicity, and bioaccumulation. A description of the integrated testing methods involved in the demonstration was presented.

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[1] U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL.

An Overview of the ARCS Program

by
David C. Cowgill[1]

Abstract

The 1987 amendments to the Clear Water Act, in Section 118(c)(3), authorize the U.S. Environmental Protection Agency's Great Lakes National Program Office (GLNPO) to conduct a 5-year study and demonstration project relating to the control and removal of toxic pollutants in the Great Lakes, with emphasis on removal of toxic pollutants from bottom sediments. Five areas were specified in the authorization as requiring priority consideration in locating and conducting demonstration projects: Saginaw Bay, Michigan; Sheboygan Harbor, Wisconsin; Grand Calumet River, Indiana; Ashtabula River, Ohio; and Buffalo River, New York. In response, GLNPO has initiated the Assessment and Remediation of Contaminated Sediments (ARCS) Program. ARCS is an integrated program for the development and testing of assessment and remedial action alternatives for contaminated sediments. The program has received in excess of $12 million dollars from 1988 through 1991. A summary of the structure of the program and the tools to be developed to address critical management questions common to many Great Lakes Areas of Concern was be presented, noting potential applicability of this work to other parts of the country.

[1] U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL.

Pilot-Scale Demonstration for the Remediation of Contaminated Sediments from the Buffalo River, Buffalo, NY

by
B. Thomas Kenna[1]

Abstract

In fall 1991, the U.S. Army Engineer District, Buffalo, in providing support to the U.S. Environmental Protection Agency's Great Lakes National Program Office, conducted a pilot-scale demonstration remediating approximately 15 cu yd of Buffalo River sediments. A thermal desorption technology was selected for demonstration because the contaminants of concern in the Buffalo River are polycyclic aromatic hydrocarbons (PAHs). In October 1991, a floating plant consisting of a barge, crane, and open clamshell bucket was used to remove the sediments from the Buffalo River and transport them in waste disposal bins to the Buffalo District's Confined Disposal Facility 4, in Buffalo Harbor. At this point, the remediation contractor, Remediation Technologies, Inc., (ReTec) of Concord, MA, took possession of the sediments and conducted pretreatment operations. These operations consisted of screening oversized material and adding water to the sediments prior to pumping into the thermal desorption unit. The sediments were treated at moisture contents ranging from 30 to 90 min. In early November, the desorption unit was demobilized to Massachusetts for completion of the demonstration, because of the significant snowfall and freezing temperatures which made completion of the demonstration in the field impractical. Extensive sampling and analyses were performed; however, results are not yet available.

The treated solids from the thermal desorption unit were solidified by the Corps of Engineers using Type I Portland cement and lake water. Four masses were prepared and cured in the field using cement-to-treating solids ratios of 0.1, 0.2, 0.4, and 0.6. Compression test results performed on cylinders prepared at the time of the solidification showed average unconfined compressive strengths ranging from 250 psi for the 0.1 cement-to-solids mass to 1,212 psi for the 0.6 cement-to-solids mass. Solidified samples were collected for eventual TCLP testing to determine leaching characteristics of the masses.

[1] U.S. Army Engineer District, Buffalo; Buffalo, NY.

Demonstration of Soil Washing Technology with Contaminated Saginaw River Sediments

by
James E. Galloway[1] and Frank L. Snitz[1]

Abstract

The U.S. Army Engineer District, Detroit, carried out a pilot-scale demonstration of a volume reduction technology in support to the U.S. Environmental Protection Agency's Assessment and Remediation of Contaminated Sediments (ARCS) program. The demonstration was conducted at the Corps' Saginaw Bay Confined Disposal Facility, Saginaw, MI, in fall 1991 and spring 1992. Approximately 300 cu yd of sediments collected from the Saginaw River (containing roughly 20 percent silts and clays and 80 percent sand, and having an average of 1.5 ppm polychlorinated biphenyls, PCBs) was processed through a grain size separation/soil washing system developed by Bergmann USA. The system consisted of a rotary trommel screen, three hydrocyclones, a dense media separator, an attrition scrubber, dewatering screens, and a flocculent clarifier. Composite samples were generated for up to 23 locations throughout the plant, using grab sample collection at approximately 1-hr intervals. Following the ARCS demonstration, a more intense monitoring of the input and output streams was conducted under the Superfund Innovative Technology Evaluation program.

The process produced four discharge streams. The majority of the feed material was recovered as washed sand with an average PCB concentration of 0.15 ppm, and an oversized fraction. Most of the PCBs were segregated into a fine fraction and a light organic fraction. The process demonstrated could produce a major reduction of the volume of dredged material requiring confinement or further treatment when the feed material is predominantly sand. One potential problem associated with the use of this and other grain size separation techniques is that the concentration of contaminants into specific fractions may generate significant volumes of more highly contaminated material requiring confinement or subsequent treatment.

The economics of handling this enriched product will play a major role in determining if a grain size separation/soil washing technique is feasible at a particular site. On the other hand, the treatment could diminish more mass of contaminants because the contaminant-enriched material lends itself to a more efficient treatment process.

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Pilot-Scale Testing of a Chemical Extraction Treatment Technology

by
Jay Semmler[1]

Abstract

Under Section 118(c)(3) of the Clean Water Act amendments of 1987, the U.S. Environmental Protection Agency's (USEPA) Great Lakes National Program Office was authorized to carry out a 5-year study and demonstration for the control and removal of toxic pollutants from the Great Lakes with emphasis on contaminated bottom sediments. The USEPA established the Assessment and Remediation of Contaminated Sediments program to implement this study.

Section 118(c)(3) identified five areas of concern on the Great Lakes to be given priority consideration for studies and demonstrations: Buffalo River, New York; Saginaw Bay, Michigan; Ashtabula River, Ohio; Grand Calumet River, Indiana; and Sheboygan Harbor, Wisconsin.

The U.S. Army Engineer District, Chicago, is tasked with management and execution of a pilot-scale demonstration of the Basic Extraction Sludge Treatment (B.E.S.T) technology with sediments from the Grand Calumet River, in northwest Indiana. This demonstration is to be completed in July 1992. The objective of the demonstration is to assess the ability of the B.E.S.T solvent extraction technology to remove (extract) organic contaminants present in the bottom sediments of the Grand Calumet River, using a patented solvent extraction technology that employs triethylamine as the solvent. Two sediment samples, with different contaminants and/or contrasting concentration levels of the same contaminants, will be tested. The first sediment sample is expected to yield relatively high levels of petroleum-based contaminants (i.e., oil and grease, polychlorinated biphenyls and polycyclic aromatic hydrocarbons) but low in metals. The second sediment sample was chosen to acquire a sample having relatively high dissolved metals and decreased organic concentration relative to the first sample.

Reporting on this demonstration will include a discussion of the results showing extraction efficiencies, residue contaminant concentrations, and cost estimates for full-scale technology applications.

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Sediment Quality Criteria: Their Utility as a Tool in Sediment Evaluation

by
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Judith C. Pennington,[1] and Victor A. McFarland[1]

Introduction

The U.S. Environmental Protection Agency (EPA) is currently authorized to develop and implement sediment quality criteria (SQC) under Section 304(a) of the Clean Water Act. SQC may be applied in many regulatory decisions, including dredged material disposal, identification of problem areas, source control, establishment of cleanup goals, development of discharge and dumping permit criteria, and determination of monitoring requirements. Therefore, SQC, when promulgated, will profoundly affect Corps dredging and disposal operations, as it is likely that aquatic disposal of dredged material and selection of disposal alternatives will be based on SQC.

Central to the development of SQC for nonpolar organic compounds is the dependence of partitioning of these compounds on sediment total organic carbon. Sediment organic carbon has been identified as the most important factor controlling partitioning of nonpolar organic contaminants between sediment and organisms (McFarland and Clarke 1986, McElroy and Means 1988) and between sediment and water (Karickhoff 1981). Many studies have also shown that partitioning of nonpolar organic compounds is strongly related to the octanol-water partitioning coefficient of the compound. Sediment concentrations expressed on a total organic carbon (TOC) basis have been used to predict concentrations of nonpolar organic compounds in organisms (McFarland and Clarke 1986; Lake, Rubinstein, and Pavignano 1987; Rubinstein et al. 1987; McElroy and Means 1988; Ferraro et al. 1990, 1991). This method is currently being pursued by the EPA to predict interstitial water concentrations for regulatory purposes (Brannon et al. 1990).

This paper presents a brief discussion of the equilibrium partitioning (EqP) approach to sediment criteria and a number of the assumptions inherent in the approach. The EqP approach has been emphasized because procedures have been developed and criteria are being formulated for nonpolar organic contaminants. The studies reported herein were designed to examine (a) the relationship between sediment organic carbon and sediment interstitial water, (b) the effects of sediment organic carbon upon bioaccumulation of polychlorinated biphenyls (PCB) and polycyclic aromatic hydrocarbons (PAH), and (c) the accuracy of the apparent preference factor as a predictive tool.

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Assumptions and Uncertainties Involving EqP

The EqP approach allows estimation of the organic carbon normalized concentration of a contaminant in sediment that will protect benthic organisms from chronic effects. The chronic water quality criterion is used as a starting point to back-calculate the organic carbon normalized contaminant concentration. Presently, the EqP approach is applicable only to nonpolar hydrophobic compounds because partitioning of such compounds between sediment and water has been shown to be related to the organic carbon content of sediments. SQC for polar organic compounds has not been developed because the mechanisms that control partitioning of such compounds between water and organic matter are poorly understood.

The EqP calculation procedure for nonpolar organic compounds is based on the relationship

$$rSQC = k_p * cWQC$$

where

$rSQC$ = sediment quality criterion, µg/kg sediment
K_p = partition coefficient for the chemical (between sediment and water), L/kg
$cWQC$ = water quality chronic criterion, µg/L

For particles with fraction organic carbon (f_{oc}) > 0.002 by weight, organic carbon appears to be the predominant sorption phase. Therefore, if $K_{oc} = K_{ow}$, then $K_p = f_{oc}K_{ow}$, where K_{ow} is the octanol-water partition coefficient and K_{oc} is the partition coefficient for sediment organic carbon (L/kg organic carbon) and is one of the key components used in EqP for predicting interstitial water concentrations.

Underlying assumptions of the equilibrium partitioning approach (Chapman 1989) are that (a) a chemical establishes equilibrium between interstitial water and sediment-associated fractions, (b) the distribution can be described by a partition coefficient between sediment organic carbon-associated contaminant and interstitial water, (c) the only toxic fraction of a sediment contaminant is that fraction freely dissolved within interstitial waters (Word et al. 1987), and (d) benthic organisms will display similar responses to contaminants in the interstitial water as did the water column organisms used to derive water quality criteria.

Materials and Methods

A sediment bioassay apparatus similar to that used by McElroy and Means (1988) was selected for the bioaccumulation studies because of its small size and simplicity. Details of the apparatus are shown in Figure 1. Two bioaccumulation studies were conducted: an initial study to develop and prove procedures, and a second study to examine the effect of sediment organic matter on bioaccumulation in greater detail. The initial study exposed clams to ^{14}C PCB-52 in Oakland Harbor sediment from Oakland, CA. Each bioassay apparatus was maintained in a water bath at 17.5 °C, the temperature at which the clams were collected. Foam plugs (McElroy and Means 1988) were used to trap PCB-52 volatilized or stripped from the water by aeration. Oakland Inner Harbor sediment was amended indirectly with either 1 or

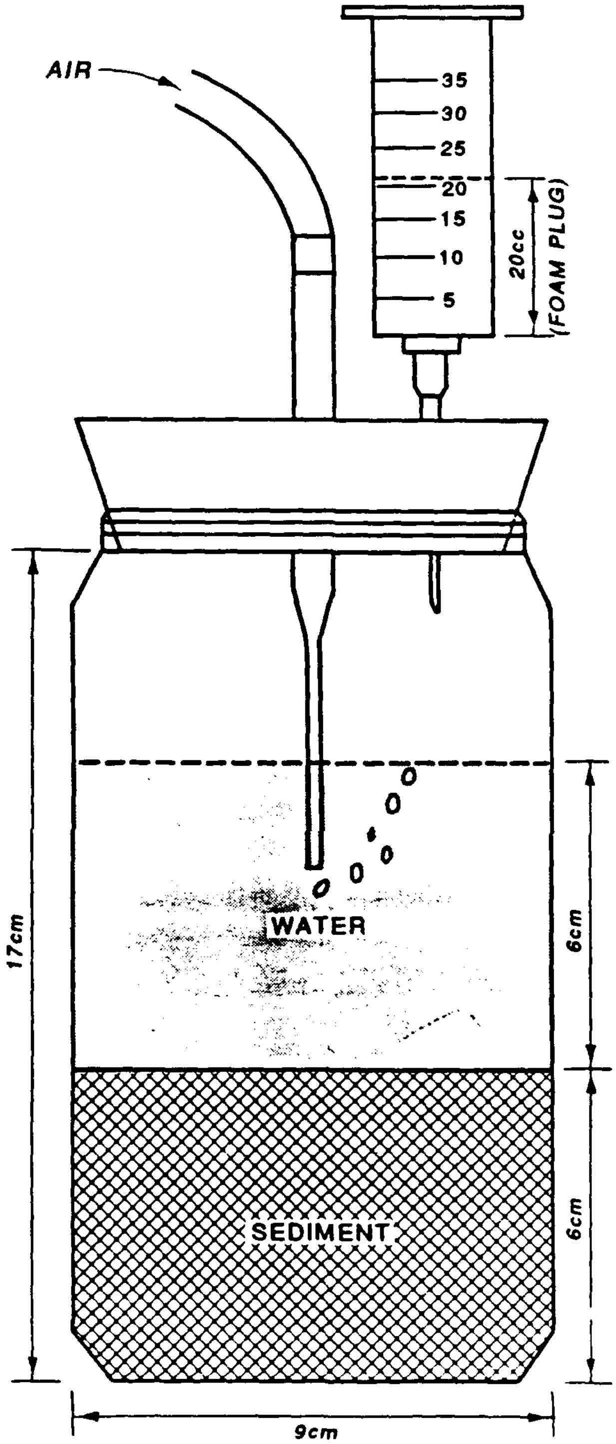

Figure 1. Testing apparatus, showing syringe body containing foam plug for collection of volatile PCBs and PAHs

10 μg PCB-52/g dry sediment. Five clams (*Macoma nasuta*) were introduced to each bioassay apparatus. Overlying water, interstitial water, foam plugs, and clams were sampled, processed, and counted by liquid scintillation (LS) at each sampling period. Clam lipids and organism dry weight were also determined. Details of the experimental procedures are presented elsewhere (Brannon et al. 1989).

In the second study, three sediments were used: Oakland Inner Harbor sediment from Oakland, CA; Red Hook sediment from the New York Bight, New Jersey; and a mixture of sediment from Brown's Lake, a freshwater lake in Vicksburg, MS, with sediment from a salt marsh channel in Louisiana. The mixed sediment provided a test of organic matter different from that in the two saline sediments (Oakland and Red Hook).

Two organisms having different feeding modes were used in this study: clams (*M. nasuta*), which burrow into and deposit-feed on surficial sediments via an incurrent siphon, and worms (*Nereis virens*), which burrow into and ingest the sediment. Clams and worms were exposed to each of the three sediments amended with 4 μg of either ^{14}C PCB-153 or ^{14}C fluoranthene/g dry sediment (described previously, Brannon et al. 1989). At all sampling periods, concentrations of PCB-153 and fluoranthene were determined in the overlying water, interstitial water, foam plugs, and clams and worms. Details of the second study are given elsewhere (Brannon et al. 1991).

All statistical analyses were conducted using methods developed by the Statistical Analysis Systems Institute (Barr et al. 1976). Analysis of variance procedures were used to test for differences between means.

Results and Discussion

Sediment organic carbon and interstitial water

Significant differences in concentrations of free and bound (complexed with dissolved organic carbon and microparticulates) fluoranthene in interstitial water were observed for sediments containing either worms or clams (Figure 2). These differences may be a function of the manner in which the organisms disturb the sediment and process carbon, or the increased bioaccumulation of organic contaminants from sediments low in organic carbon. The ability of equilibrium partitioning (EqP) to predict interstitial water PCB-153, PCB-52, and fluoranthene concentrations in sediment was tested by comparing estimated K_{oc} with measured K_{oc} values. Estimated K_{oc} values were computed by substituting values of log K_{ow} (octanol/water partition coefficient) for fluoranthene (5.5) (Tetra Tech 1985), PCB-52 (5.85), and PCB-153 (6.92) (Hawker and Connell 1988) in the equation of Karickhoff (1981) that relates K_{ow} to K_{oc}. Measured values of K_{oc} were determined by dividing the TOC normalized sediment concentration of PCB-153, PCB-52, or fluoranthene by the free interstitial water concentration of the respective compounds.

Comparison of measured and estimated K_{oc} values for the 15-day sampling (Figure 3) showed that agreement was poor for both fluoranthene and PCB-153. Similar results were observed for PCB-52. Because of the log scale of the figure and the log nature of K_{oc} values, a difference of one unit is an order of magnitude difference in partitioning between water and sediment TOC. Measured K_{oc} was consistently lower than estimated K_{oc} for PCB-153, but

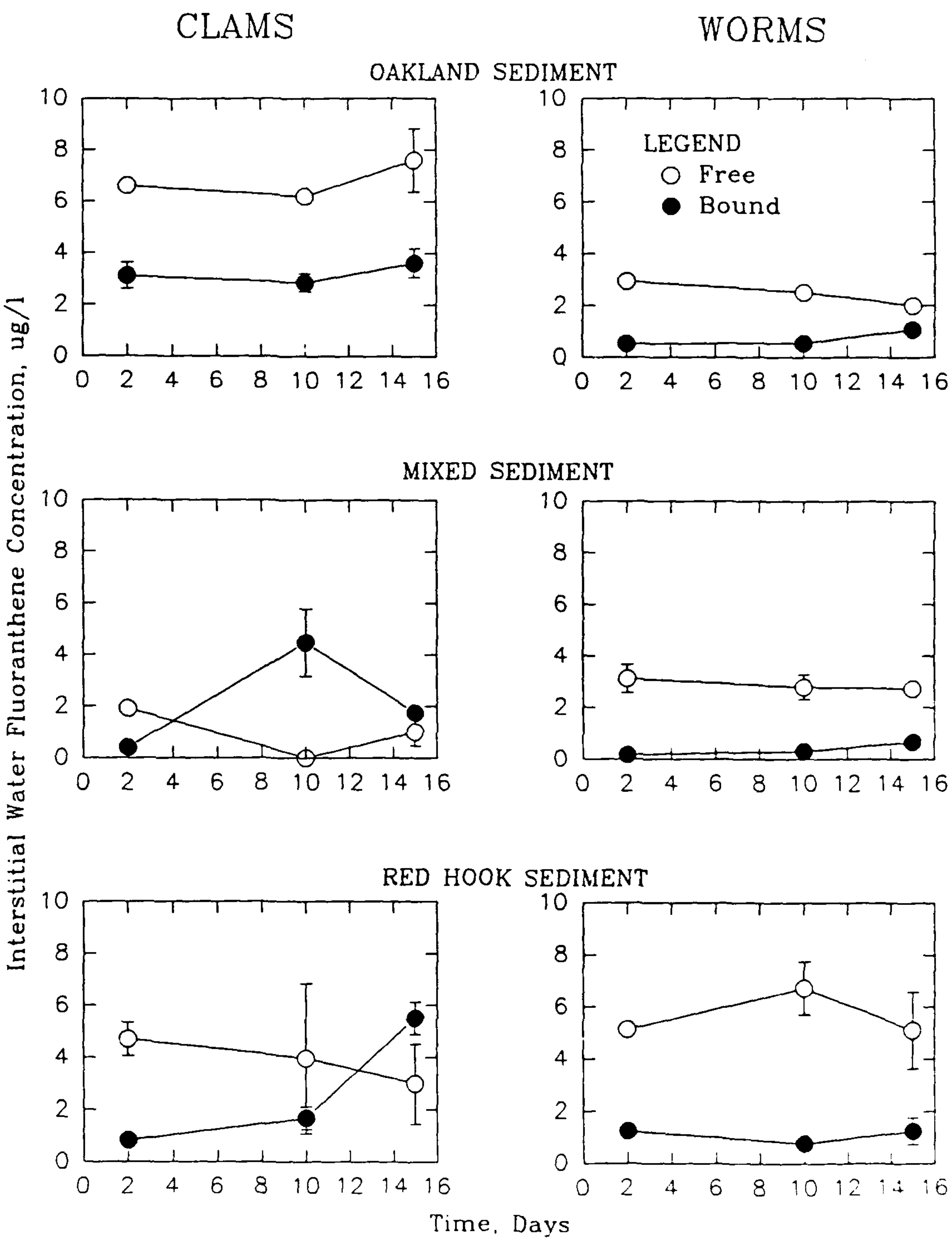

Figure 2. Free and bound (associated with dissolved TOC and microparticulates) interstitial water concentrations of ^{14}C-labeled fluoranthene during bioaccumulation testing

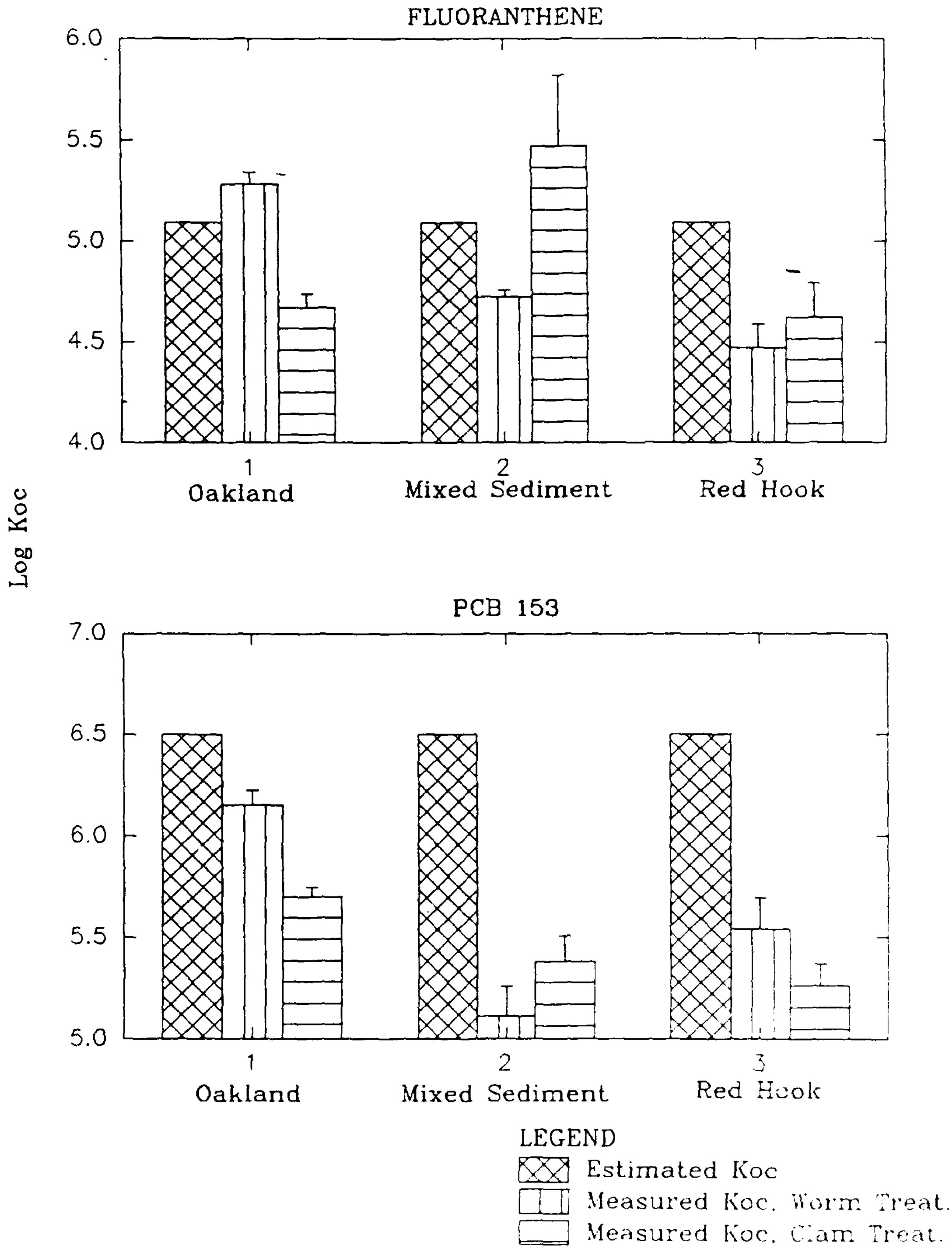

Figure 3. Estimated and measured K_{oc} values following 15 days of incubation

showed no consistent pattern for fluoranthene. Therefore. EqP did not provide accurate estimates of free interstitial water concentrations of PCB-153, fluoranthene, or PCB-52. Such inaccuracy could result in sediment categorizations that are inconsistent with the actual environmental impacts of the dredged material. In addition, such results raise questions about the ability of generalized partition coefficients based on sediment organic carbon to predict interstitial water concentrations.

Factors other than the total concentration of sediment organic carbon can affect the partitioning of nonpolar organic contaminants between sediment and water. Gauthier, Seitz, and Grant (1987) reported that K_{oc} can vary by a factor of 10 as a function of organic carbon aromatacity. Grathwohl (1990) found that K_{oc} values and K_{ow}-derived values for K_{oc} in the literature fail to account for variations in the composition of natural organic matter and are likely to be misleading. Steinberg, Pignatello, and Sawhney (1987) reported that measured concentrations in soil pore waters differ from predicted values for contaminants that have been associated with the soil for extended periods of time.

An additional problem was identified that may frequently occur in sediment from industrial areas. TOC concentrations measured using whole sediments were 1.08 percent for Oakland, 2.94 percent for the mixed sediment, and 4.63 percent for Red Hook sediment. Investigation of the Red Hook sediment revealed numerous small lumps of shiny black coal. Sorption of PCB and fluoranthene on such surfaces should be minimal in comparison to sorption on sediment organic matter because of the tremendous difference in surface area. This will result in an insignificant role of the coal fraction as a sorptive phase for fluoranthene or PCB-153 compared to other forms of sediment TOC. Passage of the sediment through a 40-mesh sieve to remove coal prior to TOC determination resulted in a 37-percent reduction in sediment TOC to 2.92 percent. This TOC concentration was then used to compute measured K_{oc} and apparent preference factors. The TOC without coal was used to generate the Red Hook data (Figure 3).

Sediment organic carbon and bioaccumulation

Tissue concentrations (μg/g wet weight) of PCB-52 increased steadily as exposure time increased in both the 1- and 10-μg PCB/g treatments (Figure 4). At the end of 23 days of exposure, clams in the 1 μg PCB/g treatment had accumulated an average of 0.35 μg PCB/g wet weight tissue, while those in the 10-μg PCB/g treatment had accumulated 4.24 μg PCB/g wet weight tissue. Some sampling times in Figure 4 have fewer than four replicates because of sample loss through death of organisms, possibly because of disease.

The relationship between sediment organic carbon and organism uptake of PCBs and PAHs was investigated by means of the apparent preference factor (APF). The APF is a measure of the preference of neutral organic contaminants for organism lipids as opposed to sediment organic carbon. The APF for each time point was calculated using the equation

$$APF = \frac{PCB_s/\%TOC}{PCB_0/\%lipid}$$

where

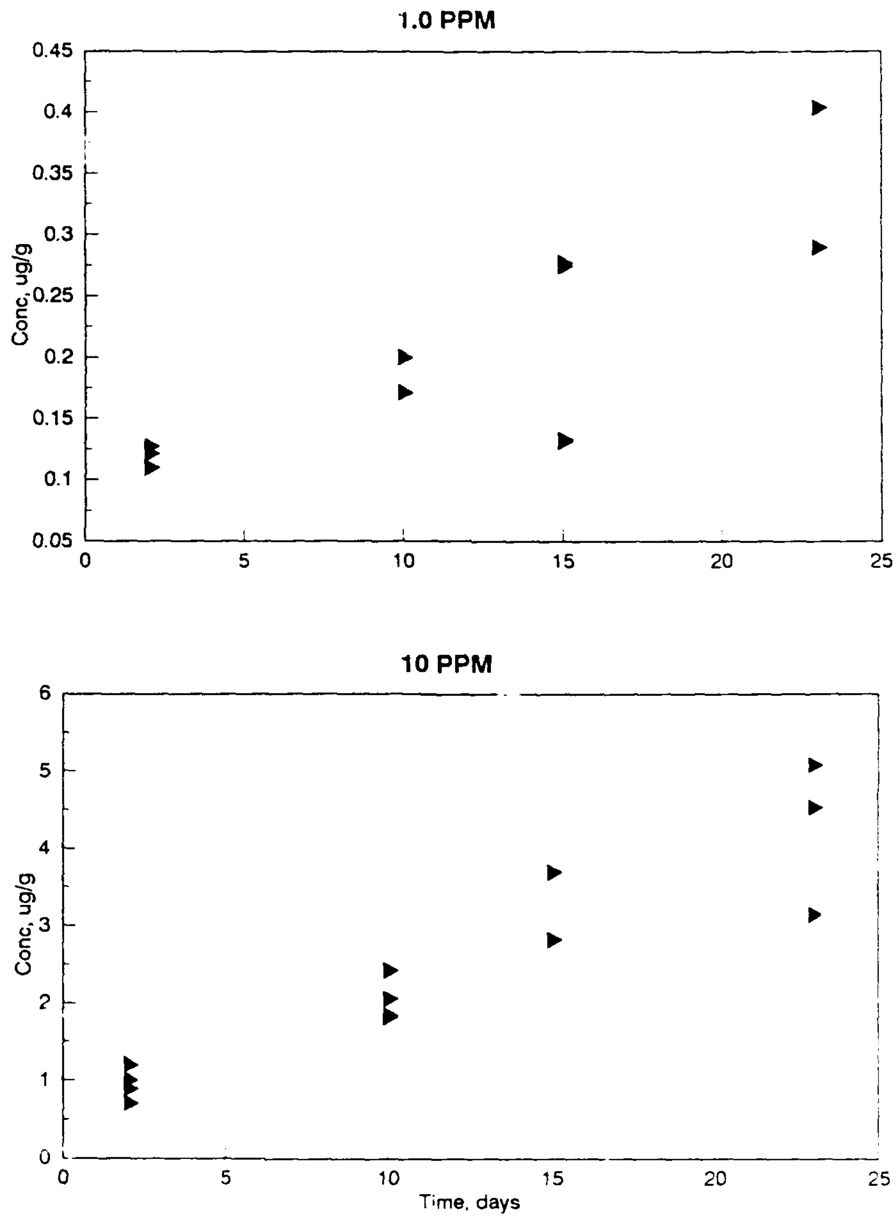

Figure 4. Replicate ^{14}C PCB-52 concentrations in clam tissue on a wet weight basis

PCB_S = PCB concentration in sediment, μg/g dry weight
$\%TOC$ = percent total organic carbon, g/g dry weight
PCB_0 = PCB concentration in clams, μg/g wet weight
$\%lipid$ = percent lipid in organism extracts, g/g wet weight

This equation, taken from McElroy and Means (1988), is based on the thermodynamic bioaccumulation potential (TBP) equation of McFarland (1984) and the preference factor equation of Lake, Rubinstein, and Pavignano (1987). TBP gives the maximum theoretical concentration of a neutral organic compound that can be bioaccumulated from sediment.

As shown in Figure 5, the APF for the 10-μg PCB/g treatment decreased as exposure time increased, although trends were not as clear in the 1-μg PCB/g treatment. The data were much less variable and the trends clearer in the 10-μg PCB/g treatment than in the 1-μg PCB/g treatment. In the 10-μg PCB/g treatment, a constant value of APF (2.3) was reached after 15 days of exposure. The APF value after 10 days of exposure in the 1-μg PCB/g treatment was 3.9, which did not statistically differ from APF values after 15 and 23 days of exposure.

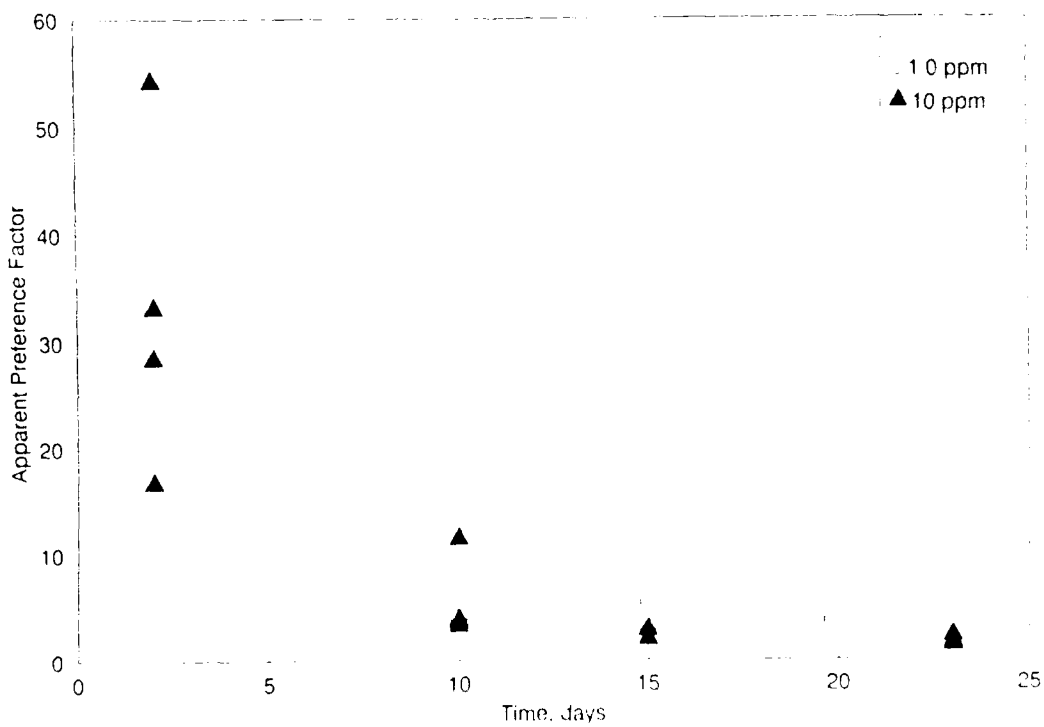

Figure 5. Replicate apparent preference factors for
^{14}C PCB-52 in clam tissue

The values of 15-day APFs for fluoranthene and PCB-153 (Table 1), as well as those for PCB-52, were similar to those for other empirical determinations reported in the literature for both field and laboratory studies and studies using both spiked and "naturally" contaminated sediment (Figure 6). The observations in this study indicate good correspondence between laboratory results using spiked sediments and results with field-contaminated sediments and biota. In addition, results of McElroy and Means (1988) and Brannon et al. (1989, 1991)

Table 1
Apparent Preference Factors (and Associated Standard Errors) for Clams and Worms Following 15 Days of Exposure to Sediment Containing Fluoranthene or PCB-153

	Clams		Worms	
Sediment	Fluoranthene	PCB	Fluoranthene	PCB
Oakland	3.77 (3.4)	4.79 (1.77)	0.8 (0.17)	4.78 (0.65)
Mixed	2.47 (0.79)	Samples lost	1.05 (0.18)	1.41 (0.29)
Red Hook	0.55 (0.84)	0.49 (0.19)	3.31 (1.27)	4.79 (2.50)

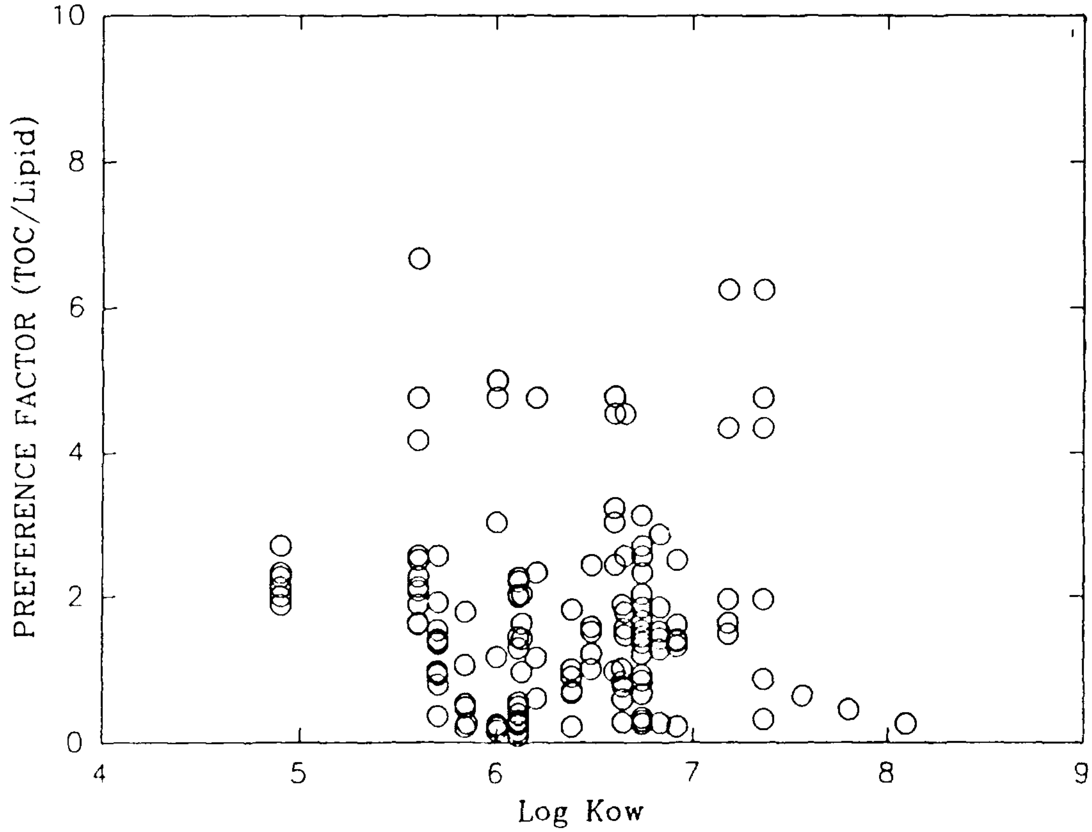

Figure 6. Literature preference factor values (reference sources: Bierman 1988; Clarke, McFarland, and Dorkin 1988; Ferraro et al. 1990, 1991; Lake, Rubinstein, and Pavignano 1987; McElroy and Means 1988; Pruell et al. 1990; Rubinstein et al. 1987)

showed rapid attainment of constant preference factors, implying that long exposures for the purposes of bioaccumulation testing are not necessary for PCBs and fluoranthene. What holds for uptake by organisms, however, will not necessarily hold for the EqP approach. The EqP approach comprises many more steps than the simple lipid:organic matter partitioning that comprises calculation of a preference factor.

Conclusions

Values of K_{oc} measured using free interstitial water concentrations of fluoranthene, PCB-52, and PCB-153 were either substantially higher or lower than estimated K_{oc} values. Our data indicated that concentrations of PCB-153 and fluoranthene in interstitial water will be either overestimated or underestimated when using equilibrium partitioning, estimated K_{oc} values, and TOC. Factors other than the total concentration of sediment organic carbon, such as organic carbon aromaticity, composition of organic matter, and the amount of time a contaminant has been associated with sediment, will also affect the value of K_{oc}. In a regulatory framework, predictive methods with a high degree of uncertainty are not a good foundation upon which to base pass/fail decisions. The geochemistry affecting interstitial water concentrations must be better understood before rigid regulatory criteria based upon predicted interstitial water concentrations are promulgated.

Bioaccumulation of PCB-52 by clams and of PCB-153 and fluoranthene by worms and clams was observed in all sediments. Even though tissue concentrations increased as time of exposure increased, APF values showed that steady state was reached between sediment-bound contaminants and organism lipid pools. No relationship was found between tissue concentrations of worms or clams and interstitial water concentrations of contaminants. This result suggests that interstitial water may not be the primary source of contaminant exposure for these sediment-associated organisms.

The APFs for PCB-153 and fluoranthene in worms and clams and for PCB-52 in clams were in close agreement with field and laboratory values reported in the literature. Our results imply that long exposures for the purposes of bioaccumulation testing are not necessary for PCBs and fluoranthene. The presence of coal in the Red Hook sediment demonstrated that care must be exercised when using TOC values for sediment from industrial areas. However, the use of sediment TOC in conjunction with partition coefficients such as APFs is a viable approach for predicting bioaccumulation of nonpolar organic contaminants by infaunal organisms.

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Considerations in Implementing Cleanup Dredging Under Section 312 of WRDA 90

by
Michael R. Palermo,[1] Tom Chase,[2] and Joe Wilson[3]

Abstract

There is a great deal of Congressional, public, and regulatory agency concern over contaminated sediments that must be dredged by the Corps to fulfill its navigation mission. Recently, the scope of such concerns has expanded to include contaminated sediments outside the confines of the navigation project, and the extent of the problem with contaminated sediments is widespread and growing. In the past, the Corps has conducted cleanup dredging and related studies for the U.S. Environmental Protection Agency under Superfund (Section 115 Clean Water Act, CWA) and is participating in Section 118 CWA activities. With the passage of the Water Resources Development Act (WRDA 90), the Corps is now authorized under Section 312 to conduct cleanup dredging under certain conditions. The provisions of Section 312 authorize the Corps to actively contribute to the goal of the CWA to restore and maintain the chemical, physical, and biological integrity of waters of the United States. The environmental benefits associated with cleanup dredging include long-term improvement of the aquatic ecosystem by reducing exposure of resources to sediment contaminants, cleanup of contamination sources to navigable waterways, improvement of water and sediment quality, and improvement of human health. This paper describes the provisions of Section 312, implications for the Corps navigation program, and considerations in implementation.

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Overview of Studies on the Adsorption/Desorption of Contaminants from Sediments in Corps of Engineers Reservoir Projects

by
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J. C. Pennington,[1] and J. M. Brannon[1]

Introduction

Management of contaminants in Corps of Engineers (CE) reservoirs is hindered by our inability to quantify contaminant movement within projects. Analytical and predictive methodologies to assess the influence of contaminated sediments in reservoirs require major development to make them usable by CE field offices (Gunnison et al. 1989). Development cannot proceed until factors influencing contaminant interactions with sediment and water are understood and procedures to measure these interactions have been established. The present work was conducted to assist with the development of these methods. This paper is based on research reported by Gunnison et al. (1991).

Objectives

Objectives of this study were to determine (a) adsorption kinetics for copper (Cu) and cadmium (Cd) at various sediment concentrations; (b) the effects of anaerobic and aerobic conditions on Cd and Cu adsorption at various sediment concentrations; and (c) PCB adsorption and desorption rates and partitioning coefficients over a range of relatively low sediment concentrations.

Materials and Methods

Sediments used in this study were collected from Green River Lake (Kentucky) and Mark Twain Lake (Missouri) by the U.S. Army Engineer Districts, Louisville and St. Louis, respectively. Batch testing to assess aerobic adsorption and desorption of Cd and Cu was conducted in Plexiglas settling columns (Figure 1) at sediment concentrations of 50, 500, and 5,000 mg/L and a metal concentration of 500 μg/L. The effect of anaerobic conditions on adsorption and desorption of Cd and Cu was determined in airtight centrifuge bottles at the sediment and metal concentrations used for the column studies.

Adsorption kinetics for a polychlorinated biphenyl (PCB-151 - [G-H^3] 2,2',3,5,5',6-Hexachlorobiphenyl) were determined by measuring uptake of 10 μg of compound at a sediment concentration of 500 mg/L. Desorption kinetics were measured by desorbing 10 μg of PCB per gram of sediment for 24 hr. Effects of sediment concentration on desorption were determined by adsorbing and then desorbing 1, 5, 10, 15, and 20 μg PCB with 250, 500, and 5,000 mg of sediment per liter. Seven desorption sequences of 10 and 20 μg PCB/g of sediment were conducted with sediment concentrations of 500 and 5,000 mg/L.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

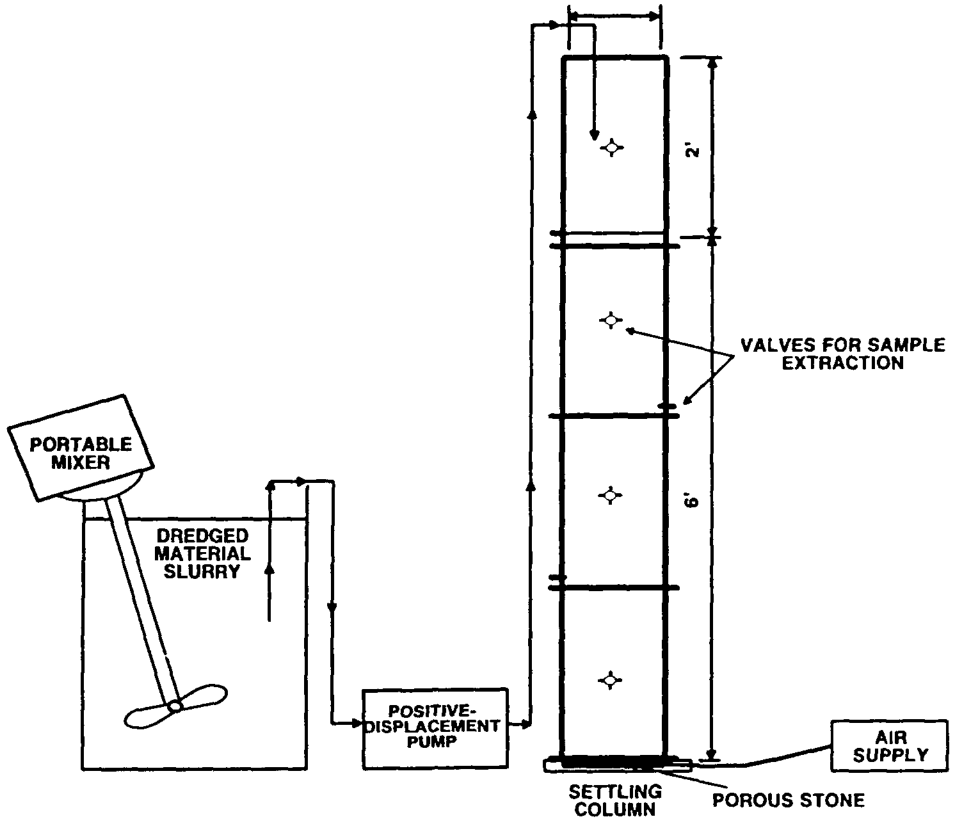

Figure 1. Plexiglas settling columns, including the mixed slurry and positive displacement pump used to load sediment into the column (after Palermo, Montgomery, and Poindexter 1978)

Results and Discussion

Cadmium and copper adsorption in Green River Lake sediment was affected by changes in sediment concentrations under aerobic conditions (Figures 2 and 3). Copper adsorption exceeded Cd adsorption at all sediment concentrations. Cadmium and Cu adsorption decreased with decreased sediment concentrations in the order 5,000 mg/L > 500 mg/L > 50 mg/L. Under initial anaerobic incubation, different sediment concentrations resulted in different concentrations of Cd and Cu in the water (Figures 4 and 5). Increasing anaerobic incubation time showed decreased differences due to sediment concentration, reaching negligible levels after 8 hr for Cd and 120 hr for Cu. Initial adsorption of added Cd and Cu was rapid under both aerobic and anaerobic conditions, nearing completion within the first 2 hr under aerobic conditions. For some treatments, the levels of Cd and Cu gradually increased as contact time increased, even under aerobic conditions.

Adsorption studies under aerobic and anaerobic conditions for Mark Twain Lake sediment showed that concentrations of Cd and Cu adsorbed under aerobic conditions decreased in the order: 5,000 > 500 > 50 mg/L (Figures 6-9). No substantial differences were found between any of the sediment conditions during 120 hr of sediment-water contact under anaerobic conditions.

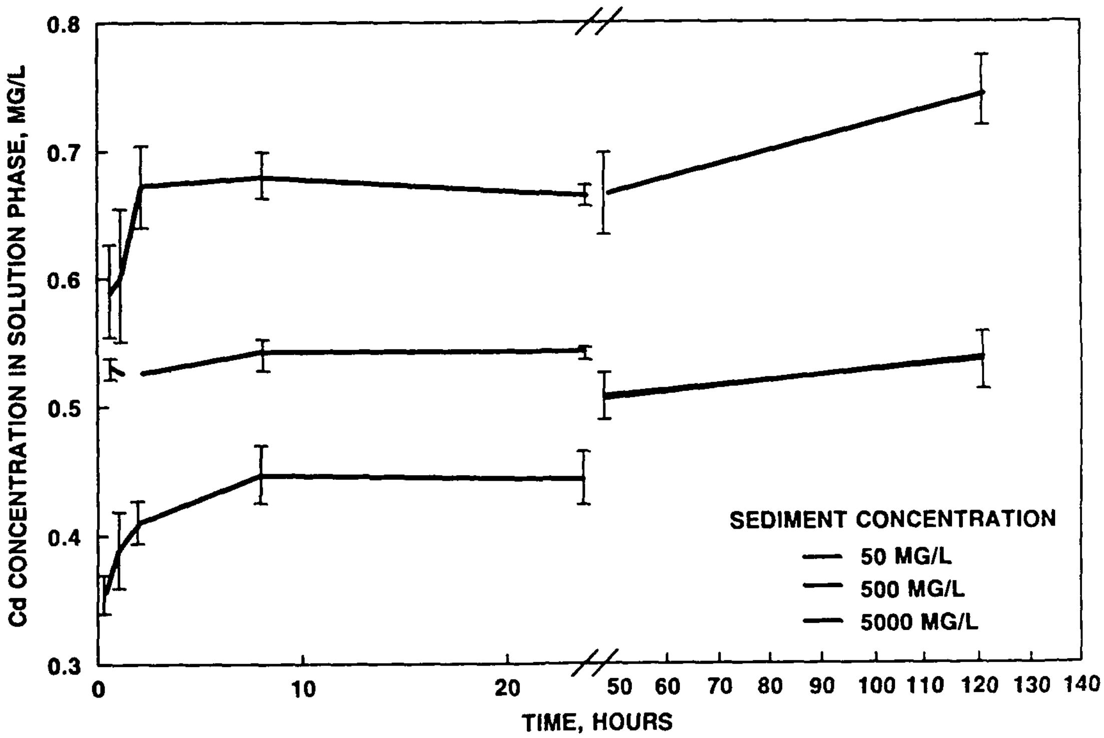

Figure 2. Effect of Green River Lake sediment concentration on adsorption and equilibrium kinetics of Cd under aerobic conditions

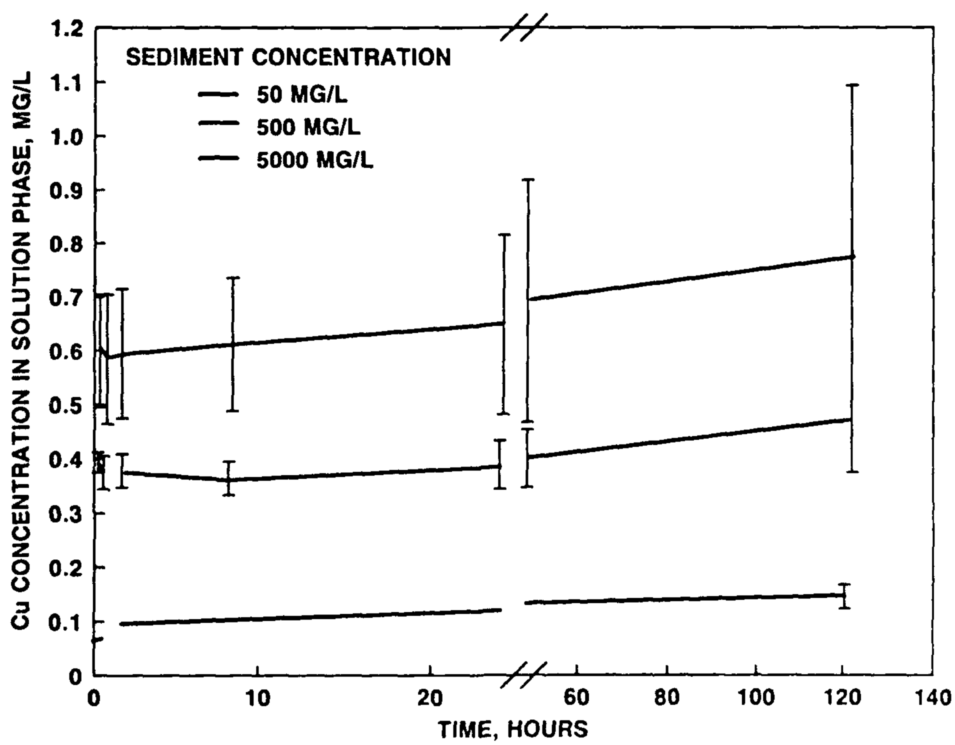

Figure 3. Effect of Green River Lake sediment concentration on adsorption and equilibrium kinetics of Cu under aerobic conditions

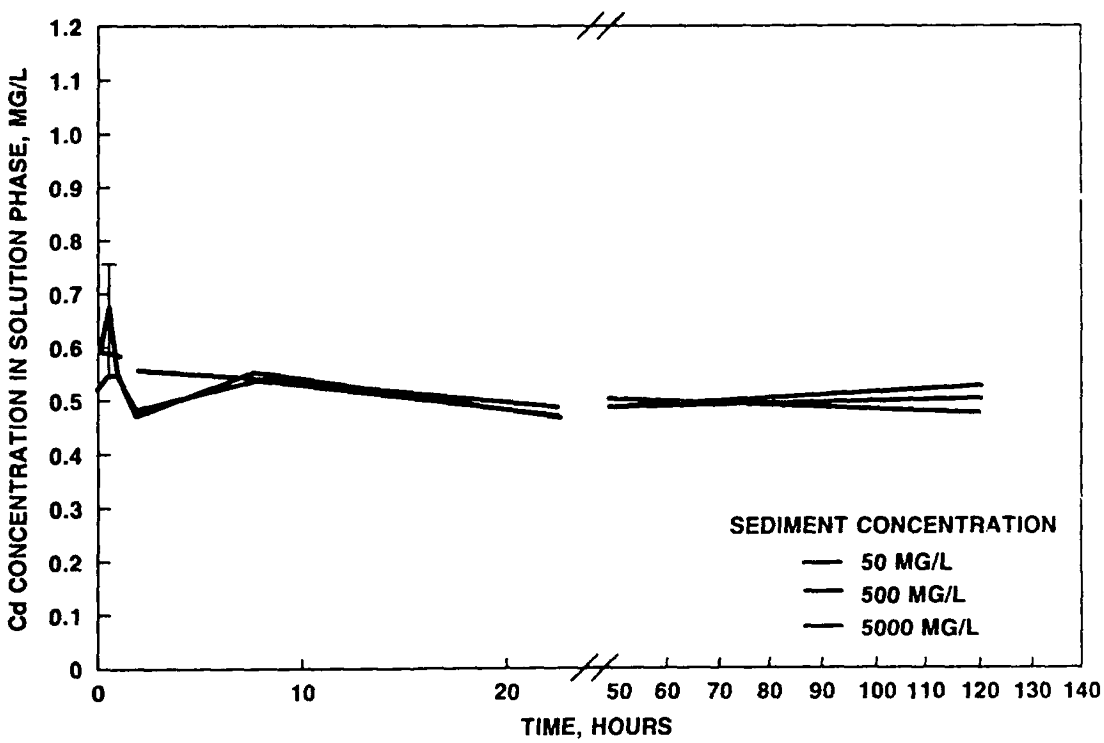

Figure 4. Effect of Green River Lake sediment concentration on adsorption and equilibrium kinetics of Cd under anaerobic conditions

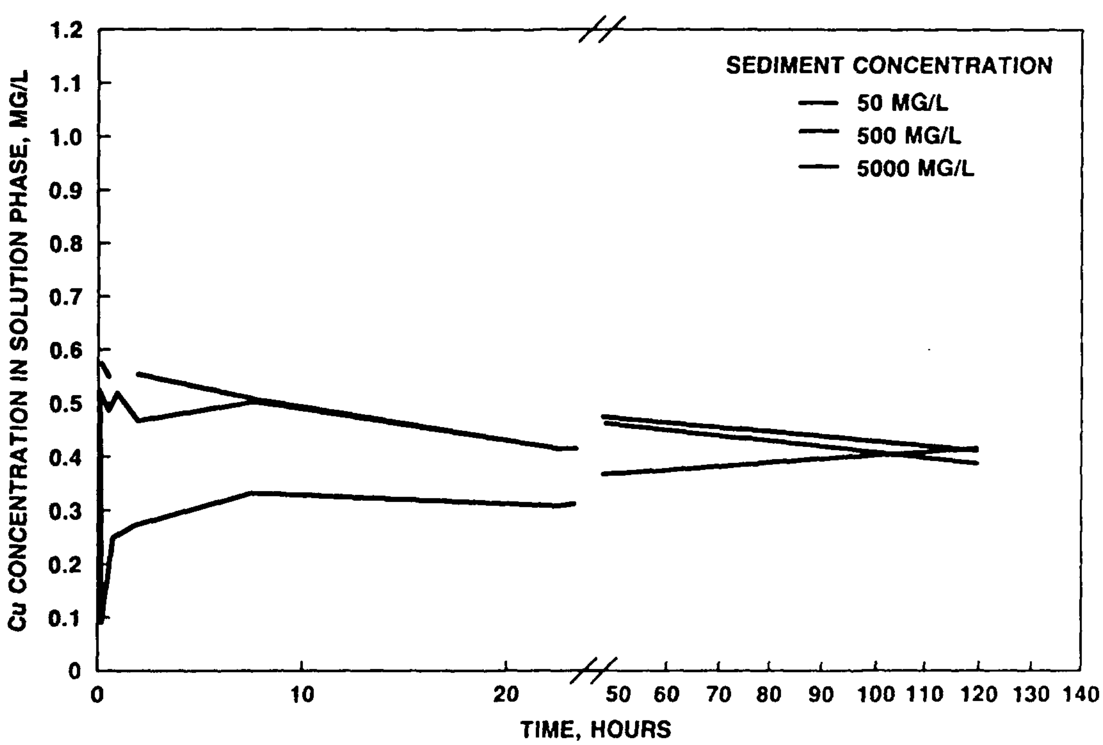

Figure 5. Effect of Green River Lake sediment concentration on adsorption and equilibrium kinetics of Cu under anaerobic conditions

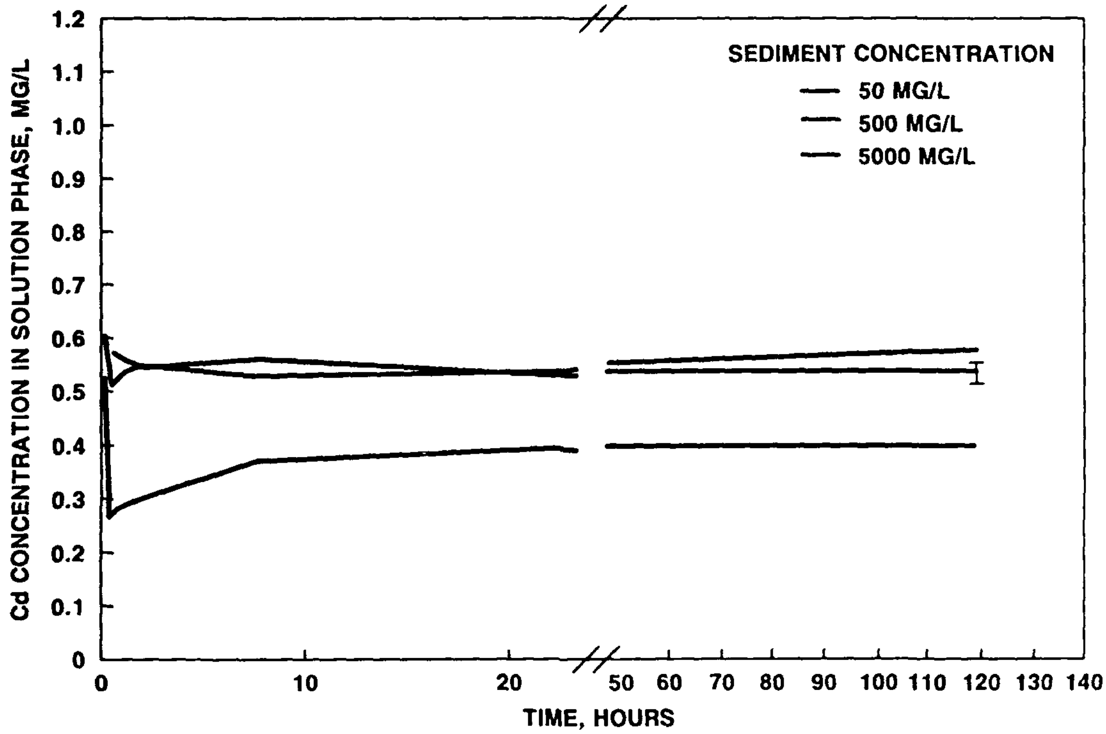

Figure 6. Effect of Mark Twain Lake sediment concentration on adsorption and equilibrium kinetics of Cd under aerobic conditions

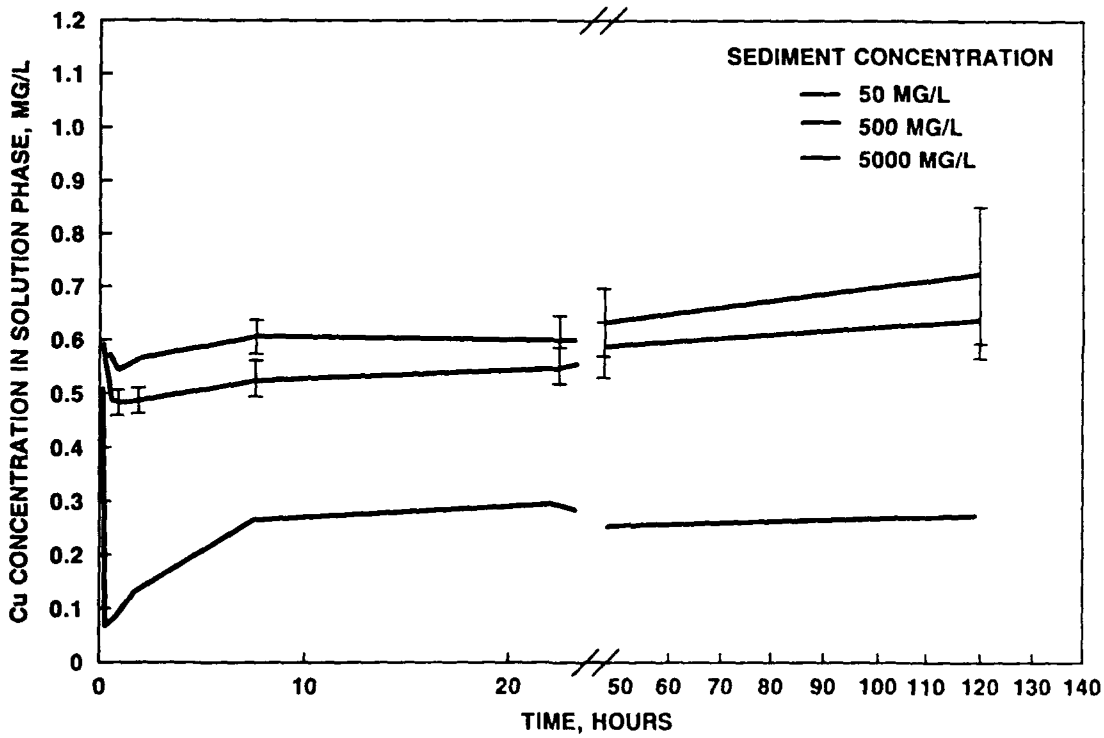

Figure 7. Effect of Mark Twain Lake sediment concentration on adsorption and equilibrium kinetics of Cu under aerobic conditions

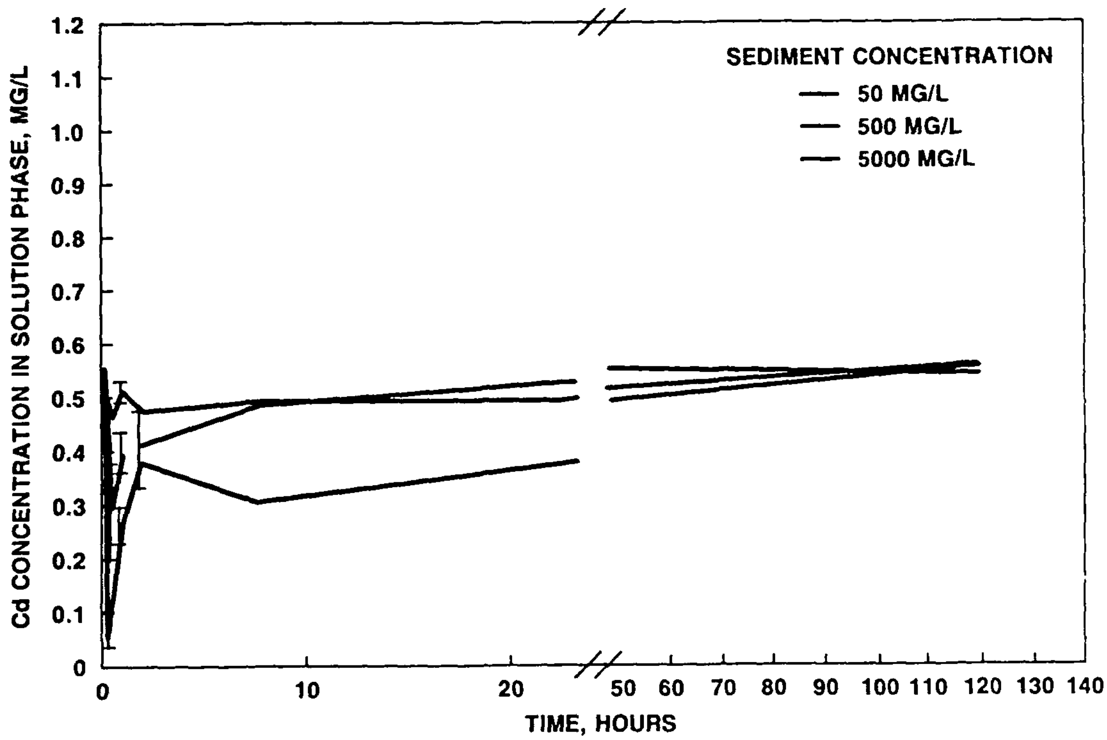

Figure 8. Effect of Mark Twain Lake sediment concentration on adsorption and kinetics of Cd under anaerobic conditions

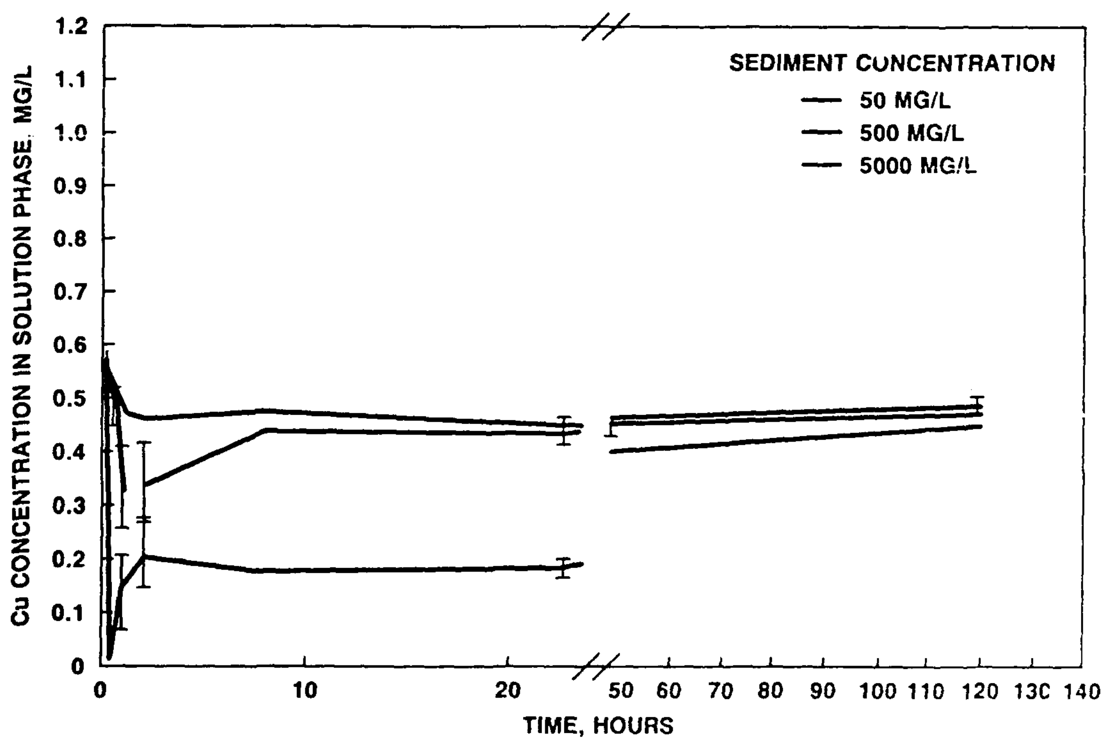

Figure 9. Effect of Mark Twain Lake sediment concentration on adsorption and equilibrium kinetics of Cu under anaerobic conditions

Cadmium and Cu concentrations in water contacting Green River and Mark Twain sediments were not affected by sediment concentrations or oxidation status of the sediment-water suspension. Desorption of Cd and Cu initially present in the sediment was minor. Adsorption kinetics under aerobic and anaerobic conditions occurred as an initial rapid adsorption, followed by a slow approach to steady state. This occurred faster under aerobic than anaerobic conditions and often was completed within 2 hr; further changes did not occur until 24 to 48 hr.

Mass-transfer steps establishing equilibrium kinetics include diffusion to the sediment, sorption by external sediment surfaces, and movement into and sorption or bonding to internal surfaces of sediment (micropores and gels) (Podoll and Mabey 1987). Diffusion of chemical to the sediment and sorption onto external surfaces are rapid in turbulent or well-mixed systems. Sorption to internal surfaces may be controlled by diffusion within the sediment pore structure (Karickhoff 1980, Freeman and Cheung 1981). The half-life of a slow sorption process is roughly an order of magnitude greater (several hours) than for sorption onto external surfaces in a well-mixed system (Karickhoff 1980). Thus, the initial rapid adsorption observed here may be the result of surface adsorption. The slower stage, which often produced an increase in Cu or Cd concentration, may be the result of physical, chemical, and/or biological processes. The reason for the increase in solution concentrations of Cd and Cu for the 50-mg/L sediment concentration is not clear, but may have been due to release of Cd and Cu originally present in the sediment.

Sediment concentration strongly influenced the magnitude of Cd and Cu sorption in aerobic treatments; levels of Cd and Cu increased at higher sediment concentrations. Higher sediment concentrations increased available sorption sites because of the larger mass and surface area/unit of mass or volume of sediment in a given volume of solution. Particle size may also affect adsorption (Singh 1971, Salim and Cooksey 1981, Podoll and Mabey 1987, Honeyman and Santschi 1988). Mark Twain and Green River sediments had similar adsorption capacities for Cd and Cu, although Green River sediment had significantly higher silt and total organic carbon content. Possible factors include silt and clay composition and quantity of oxyhydroxide compounds present.

Initial adsorption kinetics of Cd and Cu were similar under aerobic and anaerobic conditions. Factors regulating adsorption of these metals were probably not initially affected by the oxidation-reduction state of the sediments. With prolonged incubation, however, marked differences between Cd and Cu distribution in the aerobic and anaerobic sediment-water mixtures were observed. Possible causes include formation of sulfides or carbonates under increased reduction intensity and differential adsorption/desorption behavior of metals under aerobic and anaerobic conditions.

Results of adsorption kinetic tests for PCB (Figure 10) indicated that adsorption of PCB to sediments was virtually complete within 2 hr for both Green River Lake and Mark Twain sediments. This agreed with findings of previous kinetics studies (Environmental Laboratory 1987, Myers and Brannon 1988, Palermo et al. 1989), showing that 24 hr was sufficient for hydrophobic organic contaminants, such as PCBs, to attain steady-state conditions in leachate derived from dredged material. The results indicate that, even for short particle residence times, an equilibrium approach will satisfactorily describe interactions between suspended sediments and PCBs. Very little PCB desorbed from either sediment over time (Figure 11). Thus, long-term desorption will not appreciably impact test results, and the equilibrium assumption will hold for desorption as well as for adsorption tests.

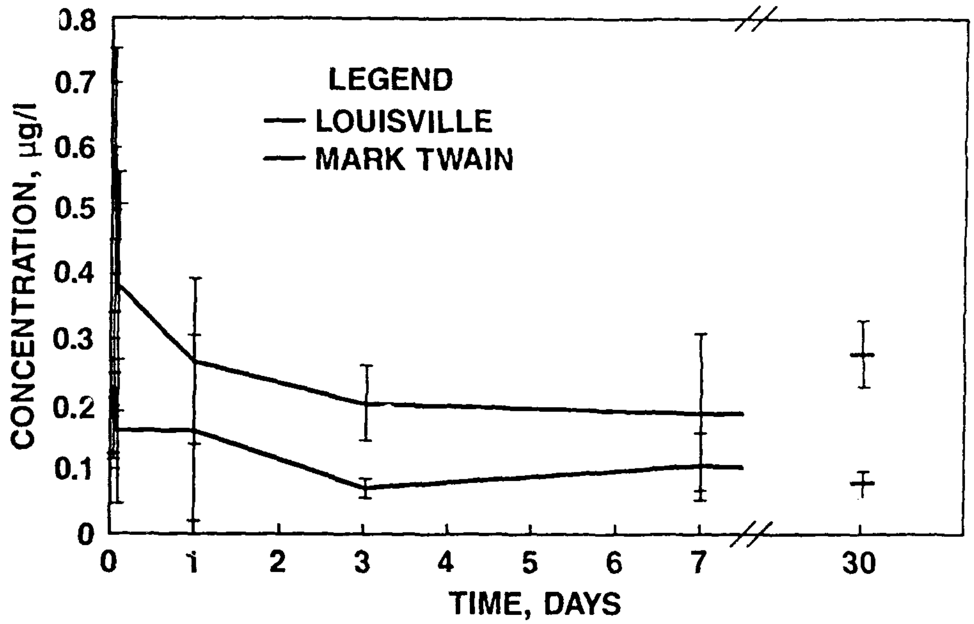

Figure 10. Kinetic curve for adsorption of PCB-151 to Green River and Mark Twain Lakes sediment

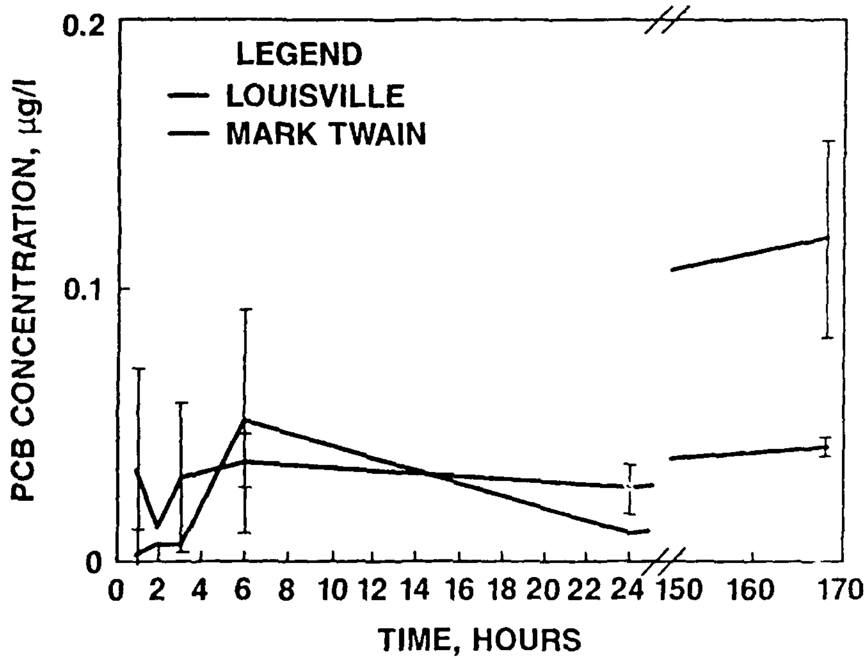

Figure 11. Kinetic curve for desorption of PCB-151 from Green River and Mark Twain Lakes sediment

PCB desorption was low at low sediment concentrations, regardless of PCB concentration in test sediments (Figure 12). At 5,000 mg sediment/L, PCB desorption increased with PCB levels in the sediment, agreeing with results reported for partitioning of hydrophobic organic contaminants (O'Connor and Connolly 1980; DiToro et al. 1982; Voice, Rice, and Weber 1983; Gschwend and Wu 1985).

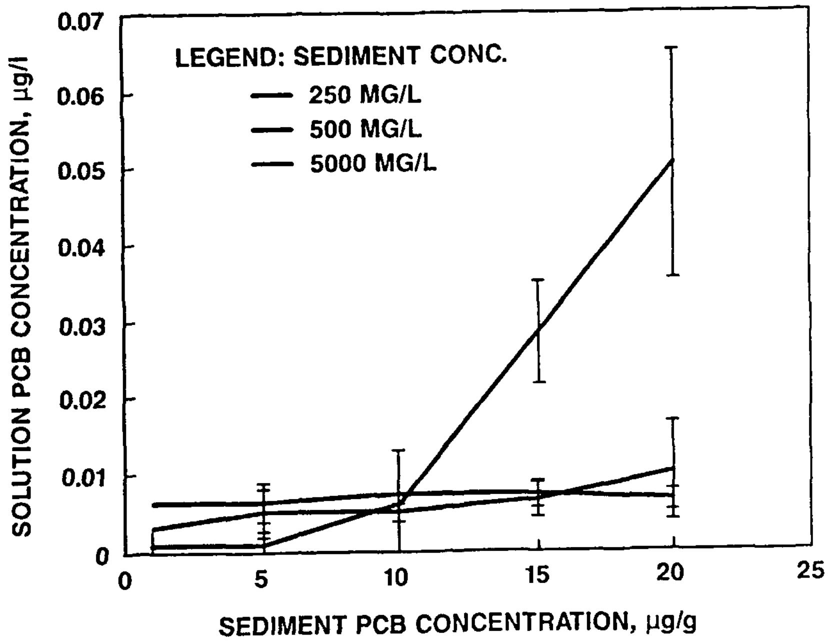

Figure 12. Desorption isotherm for PCB-151 in Green River Lake sediment

At 20 μg PCB/g of sediment, K_d decreased with increasing sediment concentration in the order 250=500 > 5,000 μg sediment/L (K_d was 3.11×10^6, 2.41×10^6, and 0.43×10^5, respectively). An inverse relationship between K_d and sediment concentration agrees with other observations (O'Connor and Connolly 1980; Horzempa and DiToro 1983; Voice, Rice and Weber 1983; Weber et al. 1983), indicating that sediment concentration is a factor in sorption tests.

Sequential desorption did not remove PCB from sediments after the first cycle, indicating that PCB is tightly bound to the sediment. Thus, the environmental fate of PCB-laden sediment will be regulated to the greatest extent by the sediment fate. Adsorption and desorption of PCB-151 to sediment occurred rapidly, permitting use of comparatively short testing times (2 hr). An inverse relationship between K_d and sediment concentration was found for PCB-151 at 5,000 μg sediment/L, suggesting an important role for sediment concentration in partitioning studies. Desorption was very limited. K_d's for low sediment concentration partition coefficients were of magnitude 10^6. The environmental fate of PCB-151 will be governed by the transport fate of the sediment to which it is bound.

Conclusions

Higher sediment concentrations increased Cd and Cu sorption as a result of increased sorption sites. Water column oxidation-reduction status did not control adsorption kinetics. Adsorption and desorption of PCB-151 occurred rapidly, but an inverse relationship existed

between K_d and sediment concentration. Extremely limited adsorption/desorption of PCB occurred at low sediment levels. The fate of PCBs strongly bound to sediment will be determined by the transport of the sediment, not desorption. For sediment concentrations of 50 to 500 mg/L, adsorption/desorption effects for Cd and Cu and for PCB-151 depended on sediment concentration.

An equilibrium approach to prediction of sediment-water interactions for metal adsorption may suffice for areas where particle residence time in the water column is greater than 1 to 2 hr. For areas having shorter residence times, a kinetic approach for adsorption may be more suitable. Results from the PCB-151 work suggest that an equilibrium approach is appropriate for describing interactions between suspended sediment and PCB, regardless of the residence time for suspended sediment.

Acknowledgments

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Short- and Long-Term Water Quality Impacts from Riverine Dredging

by
David L. Wallace[1]

Introduction

Short- and long-term water quality impacts associated with hydraulic and dragline dredging have been investigated recently by the Vicksburg District's Water Quality Section. Impacts to water quality needed to be defined and quantified in order to prepare separate Environmental Impact Statements (EIS) on three channelization projects in the Upper Yazoo River Basin (UYRB). The Dredged Material Research Program has studied hydraulic dredging operations in ocean and estuaries; however, impacts caused by dragline and hydraulic dredging of in-land riverine dredging have not been fully addressed. To evaluate water quality impacts, minor adaptations to proven numerical models and analysis of past water quality data were utilized. This paper presents the methodology and assumptions used in evaluating and quantifying water quality impacts due to dragline and hydraulic dredging channelization in the UYRB.

Water Quality Impacts

Impacts to water quality due to river dredging are both short and long term. Short-term impacts to water quality occur immediately and continue until dredging operations cease. Depending on the properties of the dredged material and the method used for dredged material disposal, impacts to water quality may extend for great distances downstream of dredging operations. The most visible impact to water quality is an increase in turbidity. The increase in turbidity is a key indicator of the resuspension of bottom sediments occurring from dredging operations. As the suspended solids loading increases, the impacts to water quality will also increase. It must be realized that during heavy rainfall, storm runoff can mask or overshadow the turbidity caused by dredging activities.

Table 1 provides a list of likely impacts to water quality as a result of riverine dredging. These impacts, which describe both short- and long-term effects, were adapted from various documents from the U.S. Fish and Wildlife Service (FWS), the U.S. Environmental Protection Agency, and previous EISs (USAE District, Vicksburg 1991). Water temperature increases are likely to occur as the result of the increase in suspended solids allowing more solar radiation to be absorbed by these particles. Since the water's dissolved oxygen (DO) content is inversely proportional to temperature, decreases in DO may occur in conjunction with increases in water temperature. The resuspension of bottom sediments will release organic carbon and nutrients, which will also decrease DO. Toxic materials such as pesticides and heavy metals, which may be attached to the sediments, may also be released.

[1] U.S. Army Engineer District, Vicksburg; Vicksburg, MS.

Table 1
Short-Term Water Quality Impacts Due to Riverine Dredging

Activity	Impact
Channel dredging	Resuspends sediments. - Increases turbidity. - Increases suspended solids. - Resuspends toxic materials. - Increases water temperatures. - Decreases dissolved oxygen. Removes aquatic habitats. - Shifts invertebrate species composition, diversity, and biomass to bottom-dwelling types. - Decreases invertebrate drift. - Diverts fish population to nongame species. Injuries, buries, and kills biota. - Lowers species richness, diversity, and biomass.
Bank revegetation removal	Removes vegetation and tree cover, and reduces bank stability. - Increases runoff, erosion, and pollution. - Increases light intensity. - Raises water temperature. - Increases primary productivity. Reduces riparian habitat. - Reduces richness and diversity of river-dependent species.
Placement of dredged material on bank	Results in runoff and sloughing of sediments into the river. - Increases sedimentation. - Increases turbidity. - Resuspends toxic materials.

Another impact due to riverine dredging is the removal of aquatic habitat areas and the burial of the surrounding fauna when the dredged material has resettled. These impacts may increase erosion, decrease fish populations, and cause a shift in invertebrate species. These impacts were not evaluated by the District but were evaluated by the Waterways Experiment Station's (WES) Aquatic Habitat Group.

Quantification of Impacts

Short-term impacts

The major short-term impact to water quality caused by dredging comes from the resuspension of bottom sediments. This results in increases in turbidity and suspended solids, likely decreases in DO, increases in temperatures, and the potential release of contaminants. These impacts were evaluated using a turbidity plume model and past water quality data.

The extent of the turbidity impacts was evaluated using the turbidity plume model reported in the Dredged Material Research Program (WES Technical Report DS-78-13, Banard 1978). This model estimates solids concentrations at various distances downstream of

hydraulic dredging operations. Six parameters are required to use the model: (1) q, the amount of material (in milligrams per liter) remaining in suspension after discharge, usually a percentage of the total discharge; (2) D, the average vertical thickness of the turbidity plume; (3) ω, the diffusion rate or horizontal spread of the plume; (4) V_s, the mean particle settling velocity of the resuspended sediment; (5) t, the time required for the mean particle size to settle a specified vertical distance, usually 10 cm; and (6) μ, the mean current velocity.

WES Technical Report DS-78-13 provides estimates and recommended values for these parameters. From these parameters, nondimensional ratios and scaling factors are calculated. Two ratios, γ and ω/μ, which are calculated from the parameters above, are used to read scaling ratios from various nomographs. From these nomographs the concentrations and distances are then determined.

Results from this model are dependent on the appropriate discharge of the dredge and the diffusion rate selected. Because of the smaller depths in the river channels compared to depths in estuaries or bays, the nondimensional parameter γ and ω/μ were not directly applicable to the model. To use the model, the nomographs were rescaled, and the appropriate scaling factors were extrapolated. The hydraulic dredge discharge was evaluated using previous dredging data. Discharges resulting from dragline dredging operations were considered to be similar to those of a small hydraulic dredge discharging a slurry with a solids content of approximately 5 percent. The dragline dredge results were determined by trial and error and compared to past dragline dredging data. Results of the model using these values indicated that the turbidity plume would extend to approximately one-half mile downstream of the dredge for dragline dredging. Turbidity plume length predictions for hydraulic dredging were less than a mile downstream.

Data collected during hydraulic dredging operations on the Yazoo River and dragline dredging operations on the Yalobusha River (Table 2) indicated that turbidity levels returned to ambient conditions within one-half mile downstream of dredging operations. The predicted values from the turbidity model were close to those measured on the Yalobusha and Yazoo Rivers. It must be pointed out that, during hydraulic dredging operations on the Yazoo River, containment areas were used. Effluent pipes from the containment areas were placed below the water surface. This reduced the depth required for the solids to settle and helped reduce the length of the turbidity plume considerably.

Data collected during dredging activities on the Tallahatchie River provided estimates on anticipated decreases in DO and increases in temperatures. Data were collected upstream of the dredge and at the effluent of a containment area. Dissolved oxygen measurements at the effluent ranged from 0.3 to 10.3 mg/L and averaged 4.9 mg/L. Upstream DO concentrations ranged from 4.6 to 12.5 mg/L and averaged 8.7 mg/L. This represents a 44-percent reduction in mean DO that occurred in the containment area. Unfortunately, DO measurements were not collected downstream of the containment area. Data collected upstream and downstream of two hydraulic dredges and the effluent return from seven containment areas indicated only slight differences in DO. The mean upstream DO was 7.1 mg/L and the mean downstream DO was 6.8 mg/L, indicating that over the dredging operation there was an average decrease of 0.3 mg/L.

These data demonstrated that while decreases in DO did occur in the containment areas, they had little effect on the river DO. These impacts are likely the result of the much smaller flows returning to the river from the containment areas, usually 1 percent of the riverflow, and the rapid mixing that occurs in the river. The likely decreases in DO were not considered to be a significant impact.

Table 2
Direct Impacts of Dredging on Turbidity[1]

Yalobusha - Dragline Dredge				Yazoo River - Hydraulic Dredge									
	1988			1980									
Distance	4 Aug	18 Aug	30 Aug	31 Jan	11 Feb	21 Feb		11 Mar		5 Jun		18 Jun	11 Jul
Ambient	43	28	130	77	68	63	71	205	200	105	100	100	101
100-200 ft	49	49	120	75	67								
		42											
400-500 ft	41	50		77	68								
		44											
750 ft				78	67	69	82	202	198	120	105	110	110
800-1,000 ft	22	36			67								
1,000 ft		25											
1,400-1,600 ft	22	24		78	66	79	67	202	199	115	105	110	
2,000 ft				77	69	67							
2,400 ft					67	69	74	201	200	110	105	98	105
1.5 mile			55										

[1] Turbidity values are expressed in nephelometric turbidity units.

Potential increases in temperature due to dredging operations were evaluated from the same data. Temperature values upstream of the dredge had a mean of 15.7 °C. Mean downstream temperatures had a mean of 16.0 °C. This represents a mean increase of 0.3 °C that occurred during the dredging operation. This slight increase in temperature was not likely to cause any significant impacts to the aquatic environment. Due to the basin's ambient high levels of turbidity, the water temperatures are already near their highest temperatures. From these data, temperature increases resulting from dredging were considered to be a minimal impact.

Another problem associated with dredging is the potential release and bioaccumulation of toxic materials in aquatic animals. With the resuspension of sediments, toxic matter attached to the sediments may become bioavailable and bioaccumulate in aquatic animals. The amount of bioaccumulation that occurs is complex and is influenced by thermodynamic, kinetic, and biological processes. A simple numerical model that calculates the theoretical bioaccumulation potential was used to evaluate these impacts. This model is completely discussed in WES Miscellaneous Paper D-91-2 (Clarke and McFarland 1991). This model provides a simple numerical method for estimating the theoretical bioaccumulation potential from sediment chemistry for neutral organic chemicals. The model assumes that concentrations of the chemical in the water are much lower than the concentration of the chemical in the sediment. If concentrations are higher in the water, then other methods for determining the potential for bioaccumulation should be used.

To use this method, the concentrations of the desired chemical and the total organic carbon (TOC) of the sediment and the lipid content in the organism in question must be known. The chemical concentrations in the sediments must be normalized by the TOC in the sediments. The normalization of the sediment concentration gives a better estimation of the amount of chemical in the sediment which is actually available to the organisms. To determine the theoretical bioaccumulation potential (TBP), only four parameters are required.

These are: (1) the sediment concentration of the chemical in question, (2) the TOC for the sediment, (3) the decimal lipid fraction or percent of the organism, and (4) the preference factor, *pf*. The *pf* is a measure of the "preference" of the neutral organic chemical for lipid over sediment organic carbon. Various methods have been reported to calculate the *pf* value, which ranges from less than 1 to more than 4. A value of 1.73 (Clarke and McFarland 1991) was used for the *pf* in the analysis.

The TBP model was used to evaluate the potential for bioaccumulation of DDT and DDE, since these were the commonly detected pollutants in the sediments. Table 3 lists minimum and maximum TBP predictions for the Steele Bayou and Tippo Bayou basins. Sediment concentrations were collected in 1990 as part of a sampling program to evaluate water quality conditions in the UYRB (Ashby et al. 1991). Fish tissue data were obtained from the FWS and from a recent report by Ford and Hill (1991). TBP predictions were calculated using the mean sediment concentrations and the minimum and maximum lipid concentrations. The TBP values were all within the range of the measured fish tissue data and well below the Food and Drug Administration (FDA) action level for DDT (5.0 ppm).

Table 3
TBP Predictions

	DDT, ppm				DDE, ppm			
Basin	Sediment	TOC	TBP	Fish	Sediment	TOC	TBP	Fish
Steele Bayou	0.0161	4,905	0.057-0.795	0.04 1.08	0.0317	4,905	0.111-1.565	0.52 8.78
Tippo Bayou	0.0015	7,950	0.003-0.046	0.05 0.22	0.0180	7,950	0.039-0.548	0.25 2.30

Comparing these data to past data (Winger, Schultz, and Johnson 1985), it was determined that the levels of pesticides in sediments and fish tissue in the Steele Bayou Basin were decreasing. For this reason, it is very likely that the rate of bioaccumulation has reached equilibrium. Although it is very difficult to quantify the actual amount of bioaccumulation that would occur from dredging, most likely it would be below FDA action limits.

Long-term impacts

To quantify long-term impacts, comparisons between sediment and water quality data were collected at five stations within a channelized reach and at five stations within an unchannelized reach on the Yazoo River. Five sampling events collected water and sediment samples that were analyzed for in situ nutrients, pesticides, and insecticides. Pesticides and insecticides were rarely detected in the sediments.

Numerical comparisons between concentrations in the channelized and unchannelized reaches were accomplished with a statistical comparison involving two means (Pennington et al. 1991). This test compared the mean concentration from five stations in the channelized reach to the mean concentration from five stations in the unchannelized reach. Separate tests were run for each sampling event. This analysis tested the assumption that mean concentrations in the channelized reach equaled mean concentrations in the unchannelized reach. If this test failed, one could assume that the mean concentrations between the two reaches were statistically different. If the test did not fail, one could assume that the mean concentrations were not significantly different. This statistical analysis, however, is not conclusive evidence

that channelization has impacted a parameter. Parameters could also be impacted by nonpoint source runoff within the reaches. The temperature, DO, conductivity, and pH values are virtually the same in both reaches. Mean values of nitrate-nitrite nitrogen, TOC, and dissolved organic carbon were statistically different in May. The November turbidity, total solids, and suspended solids mean concentrations were statistically different. All remaining tests indicated that they were not statistically different.

This study indicated that channelization had very little effect on turbidity, total solids, and suspended solids in the channelized reach of the Yazoo River, which shares similar hydrology, land use, water quality, and sediment characteristics with other rivers in the Yazoo basin.

In summary, the major impacts to water quality due to hydraulic and dragline riverine dredging operations will be the loss of aquatic habitats and increases in turbidity and suspended solids. Increases in turbidities and suspended solids resulting from dredging activities should be temporary and localized. Increases in turbidities and suspended solids will be highest during actual construction and should return to near-ambient conditions once construction ceases. These values will remain elevated until construction is completed and banks have revegetated. Increases in temperature and decreases in DO are expected to be minimal. These models and past historical data have proved very helpful in assessing water quality impacts due to riverine dredging. To help in further understanding the impacts, continued monitoring of dredging activities in the UYRB will be conducted. Monitoring along the channelized and unchannelized reaches has been extended for an additional year. These data will be used to further evaluate water quality impacts caused by riverine dredging.

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Analyzing "Less Than" Data: An Example from Sediment Bioaccumulation Testing

by
Joan U. Clarke[1]

Introduction

Dredged material disposal evaluations conducted under Section 103 of the Marine Protection, Research, and Sanctuaries Act (Public Law 92-532, as amended) often require contaminant residue analyses of sediment or organism tissue samples. Contaminant concentrations in a dredged sediment may be statistically compared with contaminant concentrations in a reference sediment. Likewise, contaminant residues in organisms exposed to the dredged sediment may be compared with those in organisms exposed to the reference sediment (bioaccumulation tests). This paper will focus on statistical analysis of bioaccumulation data, although the conclusions to be drawn can be applied as well to statistical analysis of contaminant residue data in sediment, water, or any other matrix. Statistical analyses will follow recommendations in Chapter 13 of the "Green Book" (USEPA/USACE 1991) wherever applicable.

Chemical analysis of contaminant residues frequently results in some concentrations reported by the analytical laboratory as less than a specified detection limit (DL) or statistical quantitation limit. Data that include "less-than" values represent a frequency distribution that is truncated on the left because the observations below DL (to the left of the DL on a frequency distribution graph) are unknown. Such data are termed "censored" (in this case, left-censored). Censored data present a problem for the analysis of bioaccumulation test results because they cannot be handled by routine descriptive or comparative statistical techniques. For example, calculation of a mean or variance for a data set containing unknown values is impossible. Without means and variances, hypothesis testing using parametric statistical techniques such as analysis of variance or t-tests cannot be done.

One commonly used method to avoid this problem is to simply substitute a constant (usually zero, one-half the DL, or the DL itself) for each less-than value. Another way is to eliminate less-than observations from the data analyses. However, either method can introduce substantial bias to the estimates of means and variances, and can strongly influence the outcome of statistical comparisons. Better estimates of mean and variance can often be obtained using a different type of procedure, so-called "robust" methods that combine observed data above the DL with extrapolated values below the DL.

When fewer than 50 percent of the data are less-than values, the sample median provides an unbiased estimate of central tendency, and does not require any value substitutions. Likewise, the interquartile range (IQR = 75th percentile observation minus 25th percentile observation) provides an estimate of dispersion when fewer than 25 percent of the observations are less-thans. Unlike mean or standard deviation (SD), the median and IQR are resistant to

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outliers. Nonparametric statistical techniques based on ranking of the data, such as the Wilcoxon rank-sum test (two samples) or the Kruskal-Wallis test (two or more samples), can be used for hypothesis testing. These procedures have the advantage of not requiring normality, an assumption that is often unrealistic for environmental data. However, the nonparametric tests may have less power to detect significant differences than their parametric counterparts if the assumptions of the parametric procedures are met. Severely censored data sets (more than half of the observations are less-thans) can in some cases be analyzed by first fitting a uniform distribution between zero and the DL to below-DL observations.

Techniques for analyzing censored data are illustrated and compared using an actual data set of bioaccumulation results for organisms exposed in the laboratory to a contaminated sediment and a reference sediment from the New York Bight area. This small data set exemplifies the complexity involved in deciding how best to handle less-than values. Descriptive statistics and the outcome of hypothesis testing are shown to differ considerably depending on the methods chosen to deal with less-than values. This paper does not cover adaptations of the techniques for analysis of censored data sets with multiple detection limits (i.e., more than one detection limit for a given contaminant among the replicates of a treatment).

Simple Substitution and Elimination Techniques

The data chosen to illustrate the results of techniques for handling less-than values are actual bioaccumulation data for PCB 182 (2,2',3,4,4',5,6'-heptachlorobiphenyl) in the polychaete *Nereis virens* exposed for 28 days to a contaminated dredged sediment and a relatively uncontaminated reference sediment. The bioaccumulation test was conducted according to the Green Book (USEPA/USACE 1991) specifications.[1] These data include two observations less than DL for which the analytical laboratory reported the actual concentration as well as the DL. The effect of simple substitution and elimination techniques on the data and summary statistics (mean, SD, median, and IQR) using the known concentrations ("original data") and reported DLs is shown in Table 1. Observations affected by the simple substitution and elimination techniques (i.e., those values less than DL) are printed in bold.

Figure 1 illustrates the effects of simple substitution and elimination techniques on mean and SD. Substituting zero for values less than DL will depress the mean and inflate the SD, whereas eliminating values less than DL will inflate the mean and depress the SD. The effect of half-DL and DL substitution will depend upon how close to the DL the true values of the less-thans are. In the example data set, the true values of the two less-thans are close to the DLs, and thus DL substitution has little effect on the mean and SD. The major problem of simple substitution and elimination techniques in dredged sediment evaluations is that the outcome of statistical comparisons can change depending upon the technique employed. Based on the original data, the concentration of PCB 182 in *N. virens* exposed to the dredged sediment was significantly greater than that of *N. virens* exposed to the reference sediment (one-sided Dunnett's test, $P < 0.05$) (Figure 1a). Yet substituting zero or half-DL for the two less-than values resulted in nonsignificance using Dunnett's test!

[1] The Green Book recommends analyzing bioaccumulation data by analysis of variance followed by a one-sided Dunnett's test to determine whether bioaccumulation from each dredged sediment is significantly greater than bioaccumulation from the reference sediment. However, the Green Book provides no guidelines for analyzing data that include less-than values.

Table 1
Concentrations of PCB 182 (ng/g) in *Nereis virens* Exposed to Dredged and Reference Sediments[1]

Sediment	Original Data	Substitution Technique: Zero	Substitution Technique: Half-DL	Substitution Technique: DL	Elimination
Dredged	3.20	3.20	3.20	3.20	3.20
	2.20	2.20	2.20	2.20	2.20
	2.00	2.00	2.00	2.00	2.00
	0.73[2]	**0**	**0.49**	**0.98**	
	2.30	2.30	2.30	2.30	2.30
	1.90	1.90	1.90	1.90	1.90
Mean	2.055	1.933	2.015	2.097	2.320
SD	0.797	1.054	0.878	0.716	0.517
Median	2.1	2.1	2.1	2.1	2.2
IQR	0.4	0.4	0.4	0.4	0.3
Reference	1.20	1.20	1.20	1.20	1.20
	2.00	2.00	2.00	2.00	2.00
	1.20	1.20	1.20	1.20	1.20
	0.69	**0**	**0.41**	**0.82**	
	1.00	1.00	1.00	1.00	1.00
	1.80	1.80	1.80	1.80	1.80
Mean	1.315	1.200	1.268	1.337	1.440
SD	0.494	0.704	0.572	0.463	0.434
Median	1.2	1.2	1.2	1.2	1.2
IQR	0.8	0.8	0.8	0.8	0.6

[1] Effect of substitution and elimination techniques using reported detection limits (DL = 0.98 ng/g for dredged sediment; 0.82 ng/g for reference sediment).
[2] Observations affected by substitution and elimination techniques (i.e., less than DL) are printed in bold.

Analyzing Severely Censored Data

To further illustrate the problems inherent in simple substitution and elimination techniques, the same data were used, only with a higher DL that resulted in severe censoring of the reference data (five of six values were below DL) (Table 2 and Figure 2). Again, substituting zero for values less than DL depresses the mean and inflates the SD, while elimination of less-thans has the opposite effect. These effects are dramatic for the severely censored reference data; in fact, eliminating less-thans leaves only one value (and thus, no SD). Zero substitution now produces a significant result from Dunnett's test, while elimination of less-thans produces a nonsignificant result.

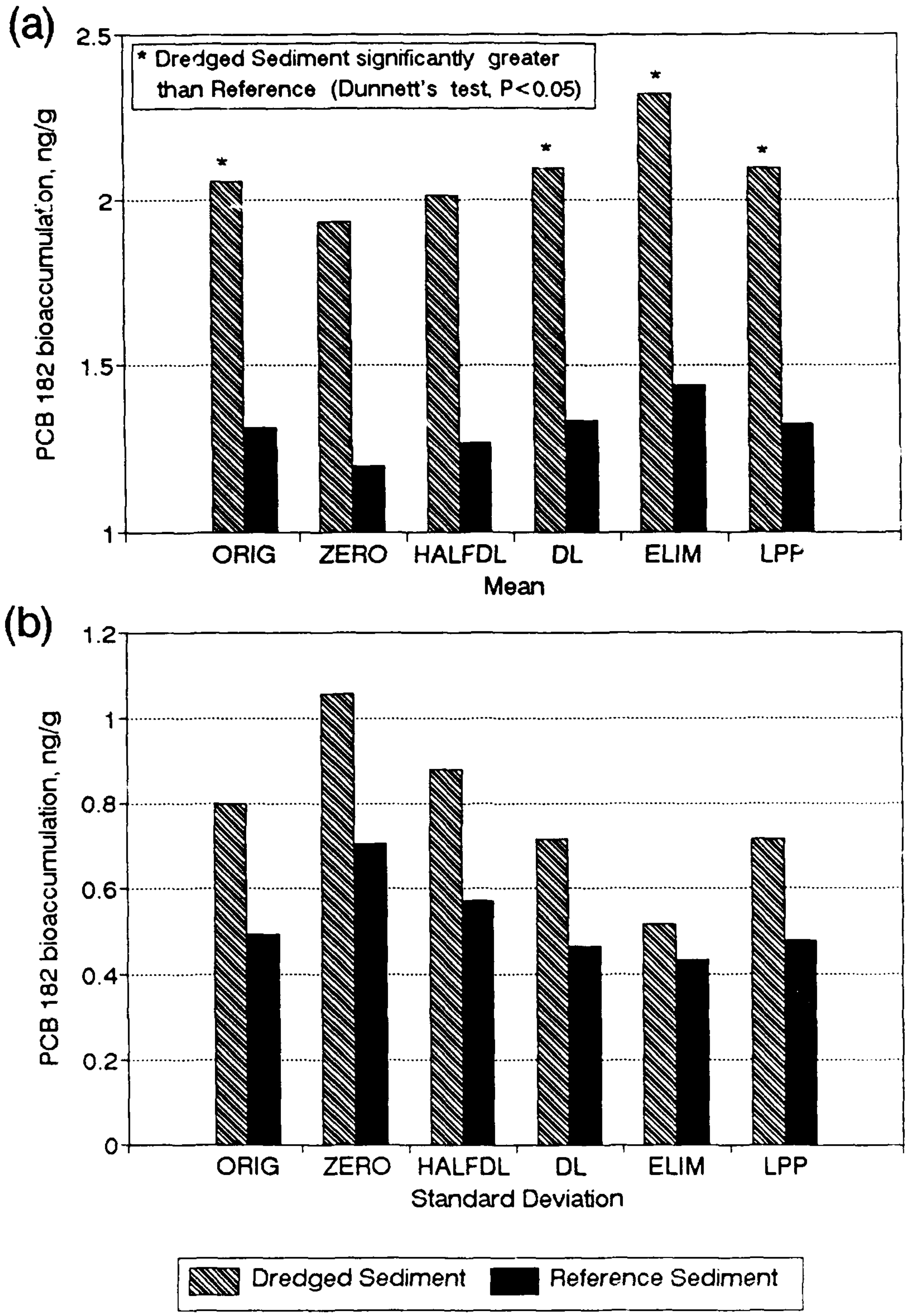

Figure 1. Effect of simple substitution, elimination, and log probability plotting techniques on PCB 182 bioaccumulation data using original detection limits (ORIG = original data, ZERO = substitution of values below DL with zero, HALFDL = substitution of values below DL with one-half DL, DL = substitution of values below DL with the DL, ELIM = elimination of values below DL, and LPP = log probability plotting technique). (a) Means; (b) Standard deviations

Table 2
Concentrations of PCB 182 (ng/g) in *Nereis virens* Exposed to Dredged and Reference Sediments[1]

Sediment	Original Data	Substitution Technique: Zero	Substitution Technique: Half-DL	Substitution Technique: DL	Elimination	Uniform
Dredged	3.20	3.20	3.20	3.20	3.20	3.20
	2.20	2.20	2.20	2.20	2.20	
	2.00	2.00	2.00	2.00	2.00	2.20
	0.73[2]	**0**	**0.95**	**1.90**		2.00
	2.30	2.30	2.30	2.30	2.30	**0.95**
	1.90	1.90	1.90	1.90	1.90	2.30
						1.90
Mean	2.055	1.933	2.092	2.250	2.320	2.092
SD	0.797	1.054	0.726	0.493	0.517	0.726
Median	2.1	2.1	2.1	2.1	2.2	2.1
IQR	0.4	0.4	0.4	0.4	0.3	0.4
Reference	**1.20**	**0**	**0.95**	**1.90**		**0**
	2.00	2.00	2.00	2.00	2.00	2.00
	1.20	**0**	**0.95**	**1.90**		**0.475**
	0.69	**0**	**0.95**	**1.90**		**0.95**
	1.00	**0**	**0.95**	**1.90**		**1.425**
	1.80	**0**	**0.95**	**1.90**		**1.90**
Mean	1.315	0.333	1.125	1.917	2.00	1.125
SD	0.494	0.817	0.429	0.041		0.797
Median	1.2	0.0	0.95	1.9	2.0	1.188
IQR	0.8	0.0	0.00	0.0	0.0	1.425

[1] Effect of substitution and elimination techniques using a high detection limit (DL = 1.9 ng/g for both sediments).
[2] Observations affected by substitution and elimination techniques (i.e., less than DL) are printed in bold.

Using hundreds of real and artificially generated water quality data sets with varying amounts of censoring, investigators (Gilliom and Helsel 1986, Helsel and Gilliom 1986) determined that a log probability plotting technique (described below) was the most robust method for estimating distributional parameters (mean, SD, median, IQR). However, this technique requires at least three uncensored observations in each data set. When a severely censored bioaccumulation data set has only one or two uncensored observations, the most suitable method may involve substitution of values from a uniform distribution for the below-DL data. The technique was described by Gilliom and Helsel (1986), who found it to be as good as the log probability plotting method for estimating distributional parameters of water quality verification data (Helsel and Gilliom 1986). The data sets were as much as 80 percent

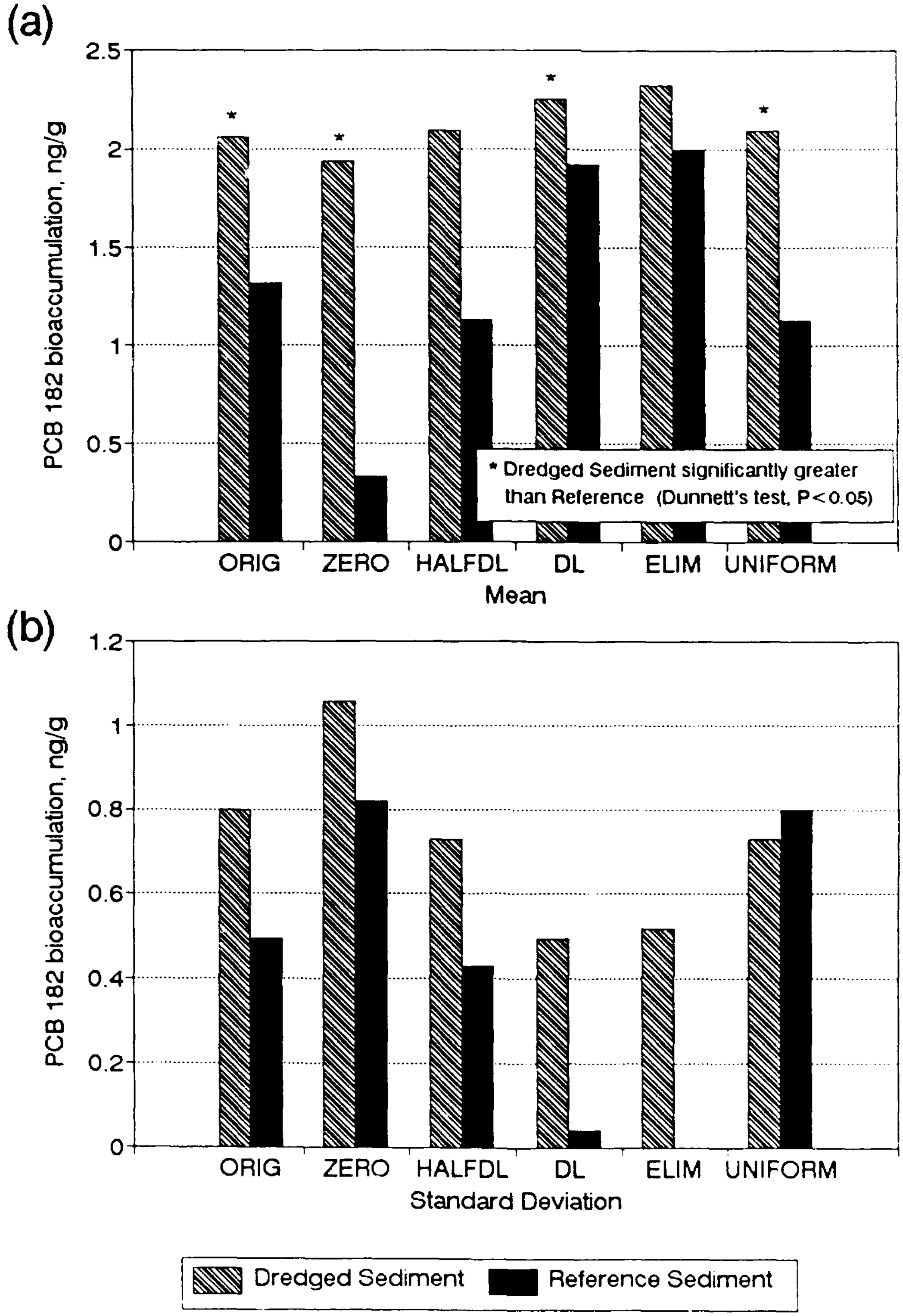

Figure 2. Effect of simple substitution, elimination, and uniform distribution substitution techniques on severely censored PCB 182 bioaccumulation data (ORIG = original data, ZERO = substitution of values below DL with zero, HALFDL = substitution of values below DL with one-half DL, DL = substitution of values below DL with the DL, ELIM = elimination of values below DL, and UNIFORM = uniform distribution substitution technique). (a) Means; (b) Standard deviations

censored. In this method, less-than values are replaced by ordered observations x_i (i = 1, 2...nc, where nc is the number of censored observations) between zero and the DL. The x_i are calculated as follows:

$$a_i = DL(I - 1)/(nc - 1)$$

This technique produces a uniform distribution for the censored data that is symmetric around one-half the DL. If only one less-than value occurs in the data set (nc = 1), x as calculated by the formula above is undefined and should be replaced by one-half DL.

The results of uniform distribution substitution on mean and SD for the high-DL (DL = 1.9 ng/g) data set are shown in Table 2 and Figure 2. Means are, by definition, the same as those obtained with the half-DL substitution technique (Figure 2a), but SD for the severely censored reference data is much higher (Figure 2b) using the uniform technique than the half-DL technique. Nevertheless, the uniform method, unlike the half-DL method, produced the same outcome from Dunnett's test as the original data (Figure 2a), i.e., bioaccumulation from the dredged sediment was significantly greater than that from the reference sediment.

One drawback of the uniform distribution substitution technique is that it cannot easily be applied to a severely censored bioaccumulation data set containing multiple detection limits.

Robust Log Probability Plotting Method

Helsel (1990) described a "robust" log probability plotting (LPP) technique for estimating distributional parameters of a data set that includes less-than values. This method assumes that the contaminant concentration data follow a lognormal distribution (i.e., the log-transformed concentrations are normally distributed). Log-transformed data above DL are regressed against their normal scores to obtain a regression equation for extrapolation of predicted below-DL values. The steps in this method are as follows:

a. For each sediment and contaminant, use all of the bioaccumulation data (less-thans should be set to slightly different values below DL) to compute normal scores.[1]

b. The log-transformed contaminant concentrations (Y) for above-DL values only are regressed against their normal scores (X).

c. The resulting regression equation is used to extrapolate predicted log concentrations for the below-DL data using their normal scores as computed in step 1.

d. The predicted below-DL estimates may then be retransformed into the original units of concentration and combined with the above-DL data to compute descriptive statistics and to perform statistical comparisons.

[1] In SAS, normal scores can be easily computed using PROC RANK with the NORMAL = BLOM option (SAS 1988).

The PCB 182 data were used to illustrate the LPP method. The original data, normal scores, and extrapolated estimates for the below-DL observations are given in Table 3. Figure 3 shows the regression lines for each sediment for $\log_{10}$ PCB 182 concentration versus normal scores, along with the actual below-DL values and their extrapolated estimates. Note that these data reveal a shortcoming of the LPP method: the extrapolated estimate for the dredged sediment less-than value is above the DL! In this case the best procedure is to substitute the DL for the extrapolated estimate since the concentration for that observation is known to be below the DL. Using the LPP method, means and SDs are quite similar to those calculated from the original data, and the results of Dunnett's test are the same (Figure 1). As

Table 3
Concentrations of PCB 182 (ng/g) in *Nereis virens* Exposed to Dredged and Reference Sediments[1]

Sediment	Original Data	Normal Scores	Above-DL Data Combined with Extrapolated Below-DL Data	Detection Limit
Dredged	3.20	1.28155	3.20	
	2.20	0.20189	2.20	
	2.00	-0.20189	2.00	
	0.73[2]	-1.28155	**0.98**[3]	0.98
	2.30	0.64335	2.30	
	1.90	-0.64335	1.90	
Mean	2.055		2.097	
SD	0.797		0.716	
Median	2.1		2.1	
IQR	0.4		0.4	
Reference	1.20	0.00000	1.20	
	2.00	1.28155	2.00	
	1.20	0.00000	1.20	
	0.69	-1.28155	**0.76**	0.82
	1.00	-0.64335	1.00	
	1.80	0.64335	1.80	
Mean	1.315		1.327	
SD	0.494		0.477	
Median	1.2		1.2	
IQR	0.8		0.8	

[1] Effect of log probability plotting (LPP) technique using reported detection limits.
[2] Observations less than DL in the original data are printed in bold.
[3] Because the estimate extrapolated using the LPP method (1.529) was above DL, the DL was substituted.

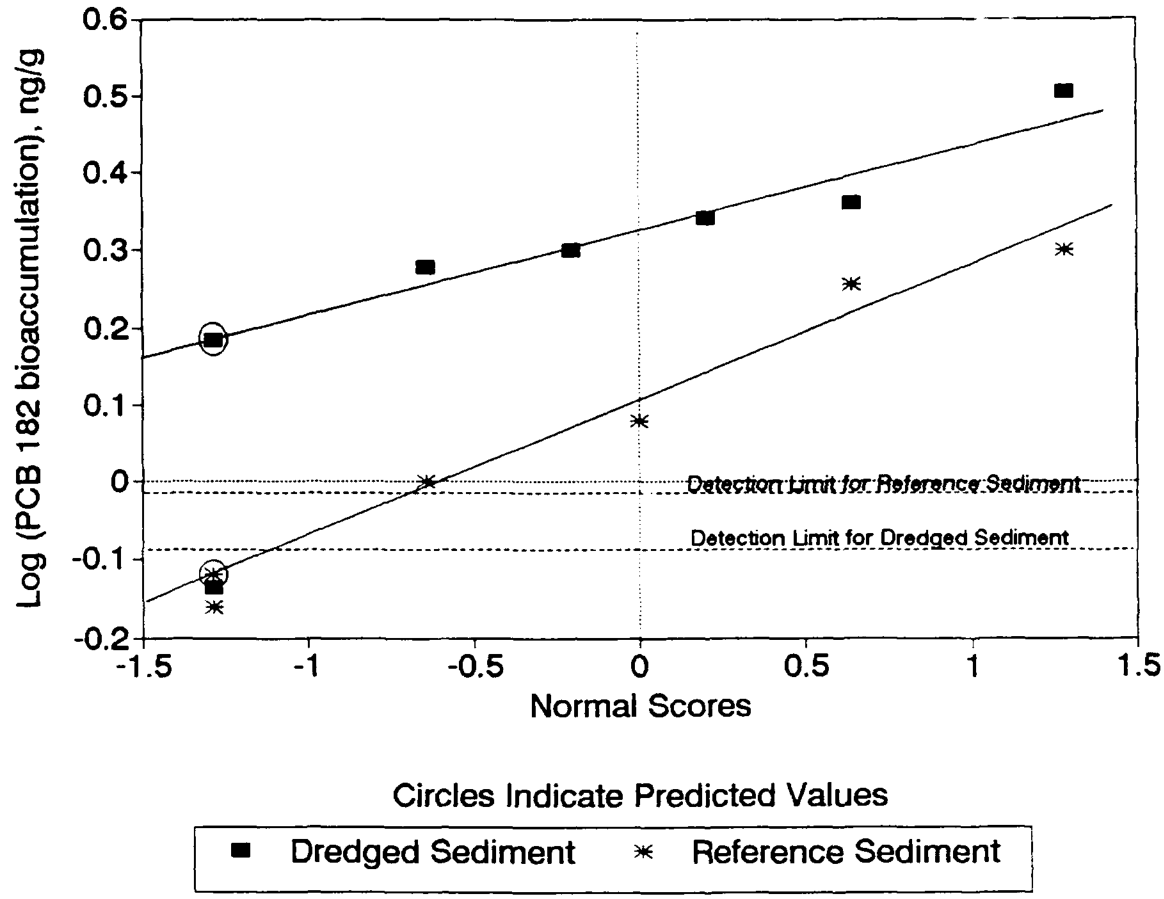

Figure 3. LPP technique: plot of $\log_{10}$ PCB 182 bioaccumulation versus normal scores

mentioned above, the LPP method requires at least three uncensored observations in a data set, and thus could not be used with many severely censored bioaccumulation data sets, such as the high-DL (1.9 ng/g) PCB 182 reference sediment data.

Nonparametric Techniques

Nonparametric statistical methods are simple and effective techniques for data analysis when fewer than half of the observations in a data set are censored. The median, or 50th percentile observation, does not change regardless of what values, from zero up to the DL, are used in place of the less-thans (Table 1, Figure 4a). The IQR is unaffected if fewer than 25 percent of the observations are censored (Table 1, Figure 4b). Note that elimination of less-thans can affect the median and IQR (Figure 4). Nonparametric tests that rank the data can be used to statistically compare contaminant concentrations among different sediments. The Green Book recommends the Kruskal-Wallis test, which is a nonparametric equivalent of

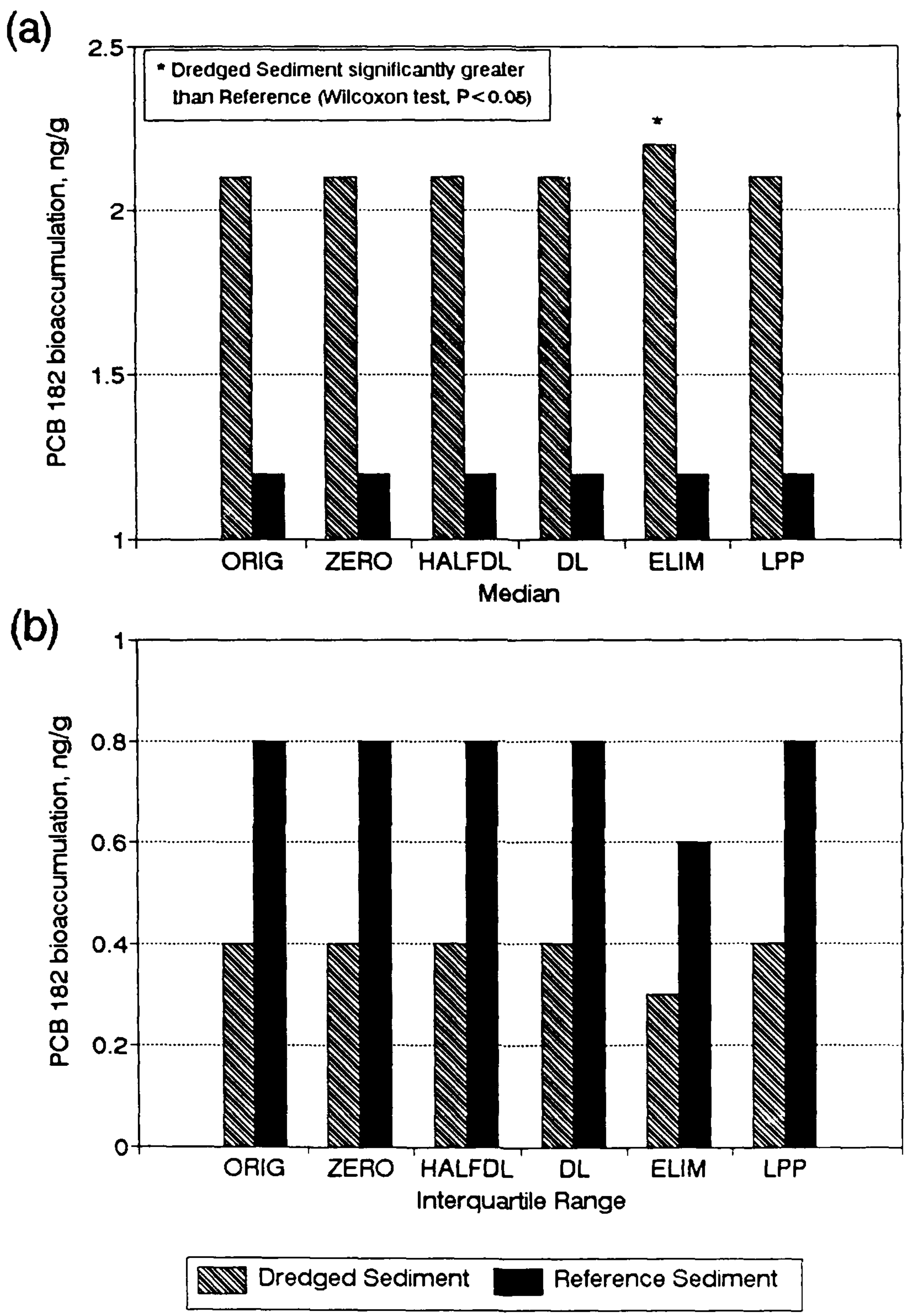

Figure 4. Effect of simple substitution, elimination, and log probability plotting techniques on PCB 182 bioaccumulation data using original detection limits (ORIG = original data, ZERO = substitution of values below DL with zero, HALFDL = substitution of values below DL with one-half DL, DL = substitution of values below DL with the DL, ELIM = elimination of values below DL, and LPP = log probability plotting technique). (a) Medians; (b) Interquartile ranges

one-way analysis of variance.[1] The Wilcoxon rank-sum test is a nonparametric equivalent of the two-sample t-test. To perform these nonparametric tests, less-thans are all assigned the same arbitrary value (e.g., zero or DL).

Using the data sets with the original DLs, bioaccumulation of PCB 182 was significantly greater in *N. virens* exposed to the dredged sediment than in *N. virens* exposed to the reference sediment only when less-thans were eliminated (Wilcoxon rank-sum test, $P < 0.05$) (Figure 4a). When the higher DL was used (Table 2), bioaccumulation of PCB 182 was significantly greater in *N. virens* exposed to the dredged sediment than in *N. virens* exposed to the reference sediment only when zero or half-DL was substituted for the less-thans (Figure 5a). When these results are compared with the Dunnett's test results (Figures 1a, 2a), the nonparametric Wilcoxon test has less power to detect significant differences in this particular case than the parametric Dunnett's test.

Severe censoring had a substantial effect upon the reference median and IQR depending on the substitution or elimination technique (Figure 5a,b). Because the lower 83 percent of the reference data (five of six observations) was censored, the median became zero when zero was substituted for the less-thans, half the DL when half-DL was substituted, the DL when DL was substituted and equal to the lone remaining observation when the less-thans were eliminated (Figure 5a). The IQR dropped to zero for the reference data when any simple substitution or elimination technique was used (Figure 5b). The uniform distribution substitution technique produced medians for both the dredged sediment data and reference sediment data that were exactly the same as those of the original data (Figure 5a), although the uniform method resulted in a grossly inflated IQR for the reference sediment as compared with the original data (Figure 5b). Nevertheless, the outcome of the Wilcoxon rank-sum test (no significant difference between dredged sediment and reference sediment bioaccumulation) was the same for the original data and the uniform distribution-substituted data (Figure 5a).

Conclusions

Chemical concentrations reported as less than DL are in most cases unknown. Thus, the regulator conducting dredged material evaluations cannot know which simple substitution or elimination technique will result in the correct outcome of statistical comparisons. In general, elimination is the worst possible alternative, and the simple substitution techniques can also result in substantial but unknown bias. Helsel and Cohn (1988) recommended that these methods be avoided. If actual values can be obtained for less-thans, they should be used in the data analysis (Porter, Ward, and Bell 1988). For data sets in which fewer than 50 percent of the observations are censored, the median is unaffected by less-thans, and nonparametric statistical comparison methods such as the Wilcoxon rank-sum test obviate the need for decisions on type of substitution technique. Nonparametric methods are more robust than parametric procedures in that they do not require assumptions about the shape of the underlying distribution, but they can also be less powerful than the corresponding parametric tests when the distributional assumptions of the parametric tests (such as normality) are met.

[1] The Kruskal-Wallis test, like analysis of variance, is a two-sided test, and cannot determine which treatments are significantly different from each other. Steel's test, the nonparametric equivalent of Dunnett's test, can identify the treatments that are significantly different from a control (Steel 1959, Day and Quinn 1989).

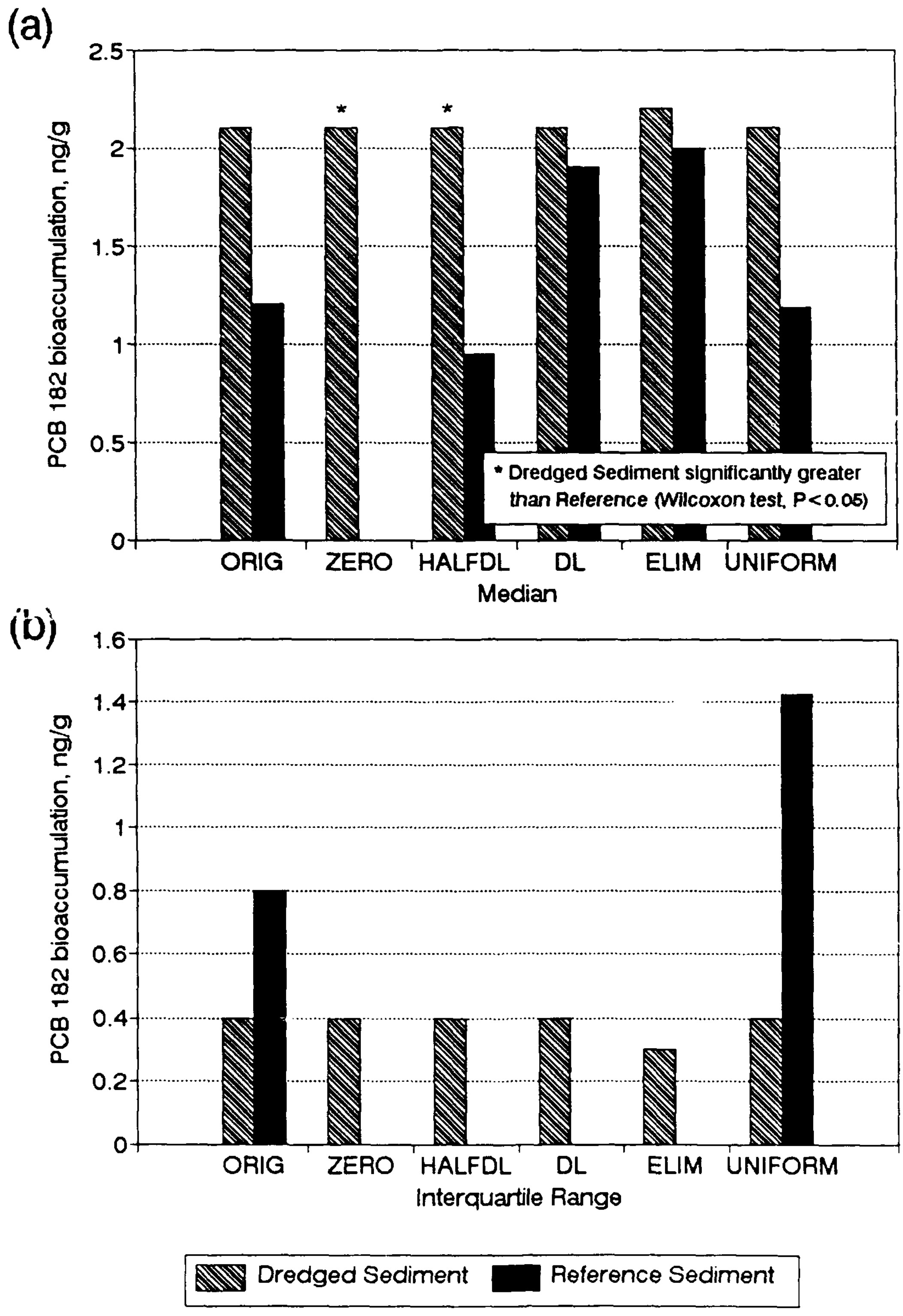

Figure 5. Effect of simple substitution, elimination, and uniform distribution substitution techniques on severely censored PCB 182 bioaccumulation data (ORIG = original data, ZERO = substitution of values below DL with zero, HALFDL = substitution of values below DL with one-half DL, DL = substitution of values below DL with the DL, ELIM = elimination of values below DL, and UNIFORM = uniform distribution substitution technique. (a) Medians; (b) Interquartile ranges

The log probability plotting method has been shown to produce good results with larger data sets (10 to 50 observations) having at least three uncensored observations. However, the general performance of the LPP method is unknown for the small bioaccumulation sample sizes recommended by the Green Book. Severely censored bioaccumulation data sets having only one or two above-DL observations may be analyzed using a uniform distribution-substitution technique.

The conclusions here, based on a single example data set, can only be tentative. In fact, statistical inference based on very small sample sizes (less than 10) is largely unexplored and perhaps presumptuous (Seaman and Jaeger 1990). Research is needed on the general performance of the techniques described herein for small (five- or six-sample) data sets before guidelines can be formulated for analyzing less-than data in dredged sediment evaluations.

Acknowledgments

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Case Study--A Dispersion Analysis of the Charleston, South Carolina, Ocean Dredged Material Disposal Site

by
Norman W. Scheffner[1] and James R. Tallent[1]

Introduction

The designated Ocean Dredged Material Disposal Site (ODMDS) for Charleston, SC, is located southeast of the jettied entrance to Charleston Harbor, shown in Figure 1. The approximately 5- by 9-km designated site has been in use for the disposal of dredged material since the mid-1970s. Historically, materials deposited at the site were predominantly fine-grained sands with some silts and clays. The proposed inner harbor deepening project will require the disposal of material of a higher percentage of fine-grained silts and clays than the original Charleston Harbor project. Therefore, an investigatory bottom video survey was undertaken by the U.S. Environmental Protection Agency (EPA) to determine whether the site contained ecologically sensitive areas that could be adversely impacted by the disposal of fine-grained material. Results of that survey identified live bottom habitat areas within the existing Charleston ODMDS.

The purpose of this study was to utilize recently obtained prototype current data to quantify the local current patterns and magnitudes, and to utilize these data to investigate the potential impact of the proposed disposal operation on the environmentally sensitive areas.

Disposal Site Analysis Procedure

A two-phased numerical modeling-based approach was used to systematically investigate the dispersive characteristics of the proposed site as a function of the local wave and current environment. The analysis procedures reported here have been developed through Technical Area 1 of the U.S. Army Corps of Engineers' Dredging Research Program (DRP) and have been applied to several site designation studies (Scheffner and Swain 1989; S.heffner, in press; Scheffner and Tallent, in press). The approach is based on the specification of local or simulated current and wave conditions to determine the short- and long-term effects of the environmental forcings on the disposal operation. The following sections describe each of these features.

Environmental forcing

A quantitative evaluation of the dispersive characteristics of a proposed or existing disposal site must be based on wave and current conditions descriptive of those actually occurring at the site. These conditions can be derived from either prototype measurements or simulated conditions that are statistically similar to prototype conditions. The methodology

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[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

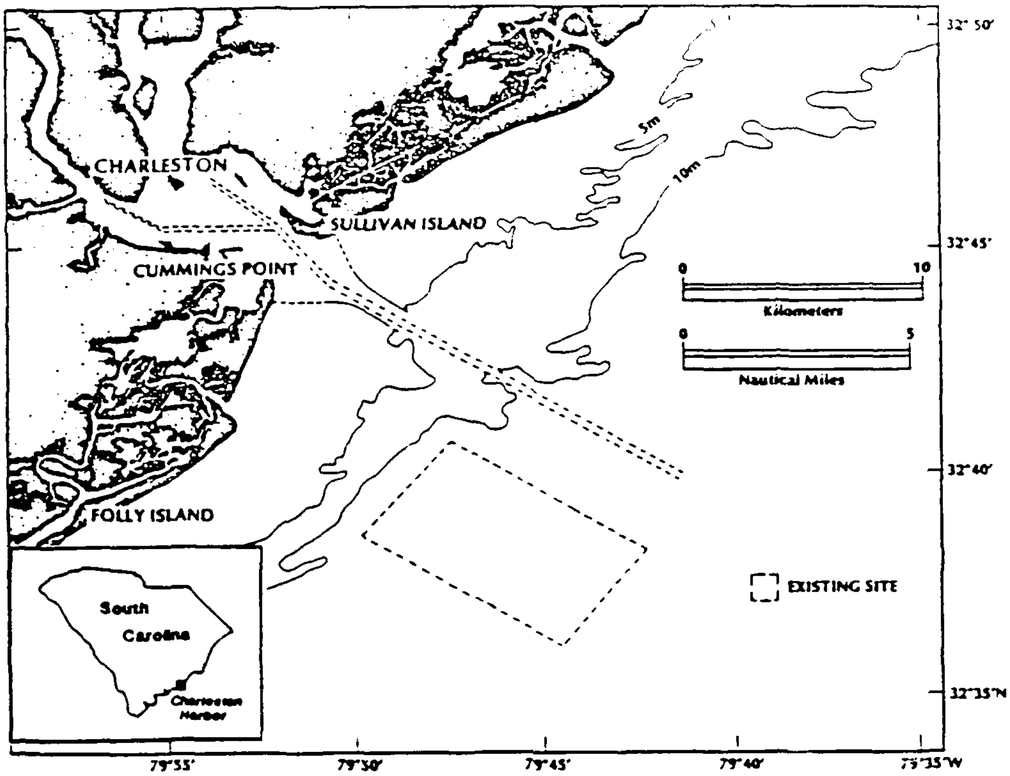

Figure 1. Charleston ODMDS location

under development by the DRP will provide a user-accessible database of site-specific wave and current information for virtually any site along the Atlantic or Pacific Ocean coasts of the United States, as well as the Gulf of Mexico and the Great Lakes. These data will include tidal height and velocity time series (except for the Great Lakes); frequency indexed storm surge height and current hydrographs; and wave height, period, and direction time series. The following sections briefly describe each of these components of the database as they apply to the Charleston area.

Wave conditions

Simulated wave height, period, and direction time series were generated according to the procedures described by Borgman and Scheffner (1991). The simulated data are based on the precomputed statistical properties of the 20-year hindcast database of the Wave Information Study (WIS, Jensen 1983); therefore, all primary statistical properties of the hindcast data are preserved in the simulation, including seasonality and wave sequencing. Hindcast data are available at a spatial density of approximately 10 to 20 miles along all coasts of the United States, including the Great Lakes. The east coast distribution of WIS data around the Charleston area is shown in Figure 2.

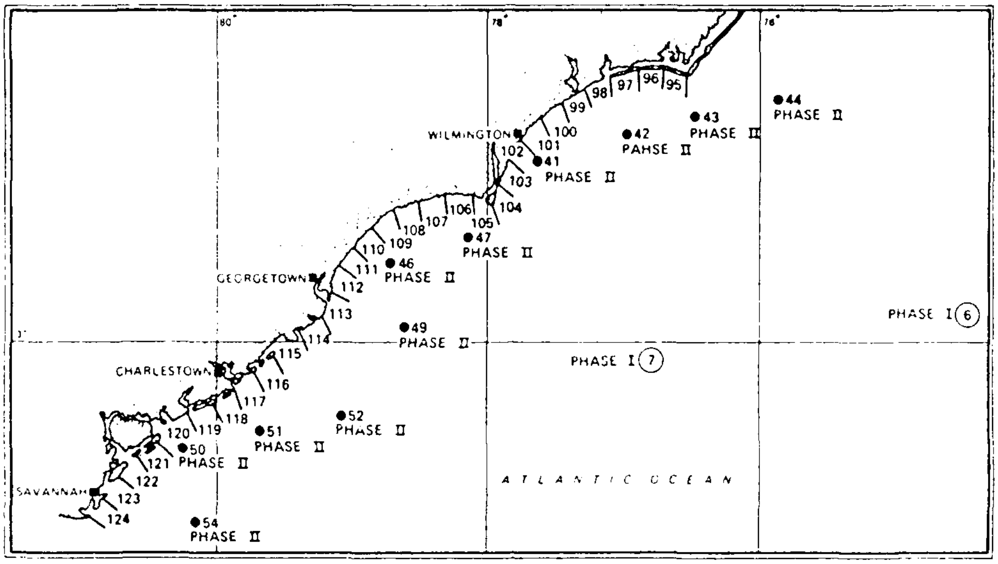

Figure 2. WIS hindcast data stations (Jensen 1983)

Tidal currents

Because the tidal simulation component of the DRP is in progress, tidal currents for the Charleston ODMDS simulation were developed from measurements taken by both the National Oceanic and Atmospheric Administration (NOAA) and the EPA. The NOAA measurements were acoustic doppler data collected at four stations in the main channel to the north of the ODMDS, two of which were located in the vicinity of the site. The EPA data consist of a single bottom-mounted current meter located in the north corner of the ODMDS. An accurate description of current magnitudes and directions at the ODMDS was determined through analyses of both NOAA and EPA data.

Harmonic analysis of the data showed that approximately 50 to 60 percent of the energy in the current time series was of tidal origin, with the tidal ellipse oriented about a northeast-southwest major axis. The primary tidal contribution is the M_2 semidiurnal component. Average tidal current amplitudes were approximately 0.66 fps. A low-frequency, non-astronomical signal component with an amplitude of approximately 0.66 fps, and a period of 6 to 8 days was also identified in the data. Mean flows reached almost 0.16 fps and were directed to the northeast.

Harmonic constituents were used to reconstruct a tidal current time series for use in the long-term simulation phase of the investigation. However, because the short-term simulations only represent conditions while the sediment descends through the water column to the ocean floor, maximum currents were selected from the analyzed data to present a worst-case situation in which sediment is transported in the water column to the live bottom area. The following two sections describe the use of the local wave and current data in the short- and long-term modeling effort.

Short-term Model

Short-term effects of the dredging operation are investigated to determine the spatial and temporal rate of change of suspended sediment concentration in the water column as the descending sediment plume disperses and is transported by ambient currents from the point of release from the barge. The modeling of this short-term phase of the operation is performed by the Disposal from an Instantaneous Dump (DIFID) numerical model (Johnson 1990). This model computes the convective descent and dynamic collapse of the sediment as a function of local current conditions, bathymetry, barge geometry, and composition of the dredged material.

Model simulations were conducted for both fine-grained (silt-clay) and coarse-grained (sand) sediment with currents specified as 1.0 and 1.5 fps. Model results include the spatial distribution of the sand and silt-clay components of the sediment load in the form of sediment concentration maps in milligrams per liter (mg/L) above ambient conditions. These maps are provided at four intermediate times, beginning with release of material and extending until sediment movement has terminated.

Results of the computations indicate that both the sand and silt-clay materials fall rapidly to the ocean floor and do not significantly impact regions beyond 0.25 mile from the release point. Computations indicate that depth-averaged sediment concentrations above background are on the order of 1.0×10^{-5} mg/L at about 1 mile from the point of disposal. Sample results presented in Figure 3 show the spatial distribution of suspended sediment concentration at approximately 30, 60, 90, and 120 min following release of material. The plot represents the approximate middepth location of 40.0 ft.

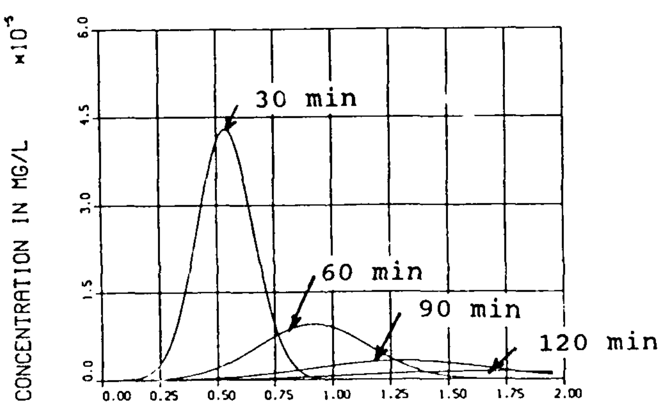

Figure 3. Suspended sediment concentration distribution at 40-ft depth

Long-term Model

The second phase of the investigation examines the stability of a sediment mound over long periods of time. This analysis focuses on the extent and probable direction in which local waves and currents erode and transport deposited material from the mound. These simulations are performed with a coupled hydrodynamic, sediment transport, and bathymetry change model that computes mound stability as a function of composition and environmental

forcings. Long-term calculations were made with an assumed 7-ft-high dredged material mound with horizontal dimensions of 1,000 by 1,000 ft. The simulated mound was assumed to be located in 37 ft of water.

Long-term simulations of disposal mound stability were based on site-specific wave and current boundary condition data. Two design sediment mounds were tested--one corresponding to a fine sand (0.100 mm) and one to a slightly finer grained sediment (0.0625 mm). The calculations are based on the equations of Ackers and White (1973) and modified to reflect the presence of waves (Bijker 1967). Simulations showed that both sediment mounds were dispersive with respect to normal wave and tidal/circulation currents and that migration rates of the centroid of the mound are on the order of 15 ft/month and 60 ft/month for the sand and fine-grained sites, respectively. Results indicate that the direction of migration is to the northeast, along the major tidal ellipse and in the direction consistent with the computed mean flow.

Because the ODMDS is located in shallow water, a storm surge hydrograph was simulated to demonstrate the effects of storm-related erosion. The simulation of a moderate-intensity event (2 fps) with a 24-hr storm surge showed the migration of the 0.0625-mm sediment mound to be approximately 155 ft. An example perspective view and contour map of the storm-deformed sediment mound is shown in Figure 4. Although storm surge-generated transport can be in any direction, computed migration rates do not indicate rapid and massive erosion, which would affect areas far removed from the mound.

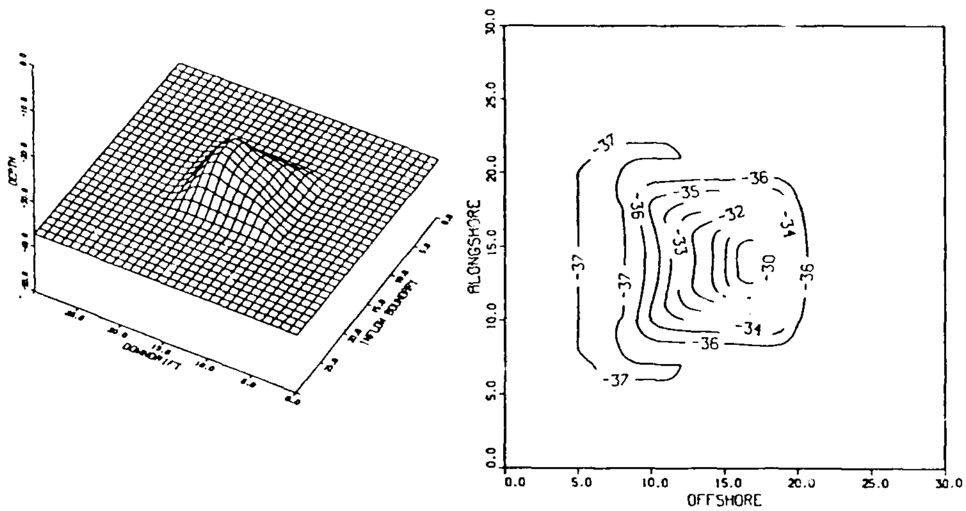

Figure 4. Storm-induced deformation of disposal mound

Conclusions

A dispersion analysis of the Charleston Ocean Dredged Material Disposal Site was conducted by the Coastal Engineering Research Center to quantify the dispersive characteristics of the site with respect to the disposal of dredged material. Of specific concern was whether the disposal operation could have an adverse effect on recently discovered live bottom communities located within and adjacent to the boundaries of the existing site.

The dispersion analysis was conducted in two phases. First, a short-term analysis was performed in which the immediate effects of the disposal operation were investigated by computing the spatial limits of sediment deposition on the ocean bottom as well as the spatial and temporal distribution of suspended sediment concentration within the water column. Second, a long-term stability analysis was conducted to determine whether local waves and currents were sufficiently severe to erode and transport significant amounts of material from the existing disposal mound onto the sensitive areas. Both short- and long-term analyses were based on numerical model predictions using site-specific wave and current boundary conditions developed through analyses of existing data.

Conclusions of the investigation showed the site to be moderately dispersive, with the primary direction of dispersion and erosion being to the northeast, although storm-induced erosion could be in any direction. Computations presented herein provide guidance on the dispersive characteristics of the disposal operation. This guidance can be used to help optimize use of the site as well as to minimize any possible adverse impact to the live bottom habitat area.

Acknowledgments

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San Francisco Bay Dredged Material Disposal Management

by
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Introduction

The U.S. Army Engineer (USAE) District, San Francisco, has the Congressionally authorized mission of maintaining and improving the navigable waters in northern California, including San Francisco Bay. Annually, more than $5.4 billion of economic activity is directly dependent on the deep- and shallow-draft navigation channels of the San Francisco Bay region.[2] The region has an annual disposal requirement of approximately 8 million cubic yards (mcy) to maintain these channels. The region also has an one-time new work requirement of approximately 19 mcy for the Oakland Harbor (7 mcy), Richmond Harbor (1.5 mcy), John F. Baldwin Phase III Navigation Channel (9 mcy), and two Navy projects (1.5 mcy). Insufficient capacity exists with present regional disposal options to accommodate these maintenance and new work projects. Nationwide disposal sites have become increasingly difficult to identify because of limited existing capacity, lack of new locations, or environmental concerns. Over the last decade, the San Francisco Bay region has been faced with disposal site uncertainty because of all three of these conditions.

In 1984, the District initiated a technical studies program, called the Disposal Management Program (DMP), to focus in-house effort in an efficient and effective manner to resolve the local disposal issue. Many events transpired over the next several years that hindered the establishment of a final dredged material disposal management plan for the region (Wakeman, Chase, and Roberts 1990). These events generated new environmental issues and public interest in the disposal of dredged material in the estuary. Because the DMP had to address these issues and respond to public concern in order to successfully complete its task, a new approach was sought. The approach necessitated a renewed need to build consensus, fully identify all feasible disposal options, and develop a long-term, implementable management plan for their utilization.

The program was reconfigured in 1990 to fit into the Corps' Long-Term Management Strategy (LTMS) approach. A LTMS management structure has been organized and is requiring consensus between the involved parties at the executive, management, and working levels to define and conduct the program--its work and its financing. This paper describes the

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[1] U.S. Army Engineer District, San Francisco; San Francisco, CA.

[2] Beeman & Associates, Inc. 1990. Benefits related to navigation channel maintenance. Prepared for U.S. Army Engineer District, San Francisco, under contract with Manalytics, Inc., Portland, OR.

developing techniques for managing the technical aspects of the LTMS program and the consensus-building process necessary for successful completion of the multiple-participant LTMS program.

LTMS Formation

In spring 1989, a comprehensive technical studies program (Plan of Action dated June 1989) was developed by the San Francisco District with the cooperation of the U.S. Environmental Protection Agency (EPA), Region IX. The program's objectives were to identify a full array of disposal options and to address regional environmental concerns under the District's DMP. The estimated technical program cost was $8.9 million plus $800,000 in Corps staff cost. In May 1989, the South Pacific Division proposed this comprehensive program to Corps Headquarters. In September, the Division was advised by Headquarters that the study effort should be reduced and limited to the Federal navigation baseline study requirements and have a cost similar to studies elsewhere ($5 million). Furthermore, the guidance stated that the Corps should not be required to carry the full financial burden of the DMP. Headquarters proposed that cost sharing be pursued in accordance with the following distribution of estimated costs: 50 percent Corps, 20 percent US Navy, 10 percent EPA, and the remaining 20 percent contributed by non-Federal navigation interests. The Division was instructed to develop a revised plan in coordination with these cost-sharing partners.

In November 1989, the Division Commander presented the Headquarter's guidance to the members of the Federal Interagency Steering Committee. Several members, particularly the Regional Administrator, EPA, and the local Commander, US Navy, had serious concerns that the proposed limit in the proposed study funding would place it at grave risk of producing an unacceptable product because of unaddressed regional technical issues. The Committee suggested that all state agencies, major Bay Area ports, key development interests, and environmental organizations be apprised of plans to pursue a limited study and of the new requirements for both Federal and non-Federal cost sharing. A meeting of all potentially interested parties was held in December. The participants were briefed and invited to comment on the proposed Corps action within the next several weeks.

On January 25, 1990, a second meeting was convened to receive comments from the involved constituencies regarding their recommendations and capabilities to support a regional program to develop a dredged material disposal management strategy. The participants stated that if they, the State, and other local navigation interests were going to be asked to contribute funds for the program, their desire was for the program to undergo a local developmental process and for them to have input into the decisions involving utilization of study funds. Furthermore, this type of participation would be critical to gaining the ultimate acceptability of the program's results from both the other Federal as well as non-Federal constituencies. The outcome of that meeting was that a need existed for a long-term management strategy that involved state, ports and development interests, environmental groups, and fishermen's associations. It was agreed that the program should be developed locally and that additional resources beyond the proposed funding ceiling would be required if the studies were to be credible and adequately address regional concerns.

The proposed approach necessitated a renewed need to build consensus, to fully identify all technically feasible and environmentally acceptable disposal options, and to develop a study program leading to a long-term, implementable management plan for their utilization. The program was tailored to fit the Corps' LTMS approach (Francingues and Mathis 1989)

and was renamed accordingly. The scope of the program was expanded to encompass all present and future activities related to the placement of dredged material in the San Francisco Bay region for the next 50 years.

Management Structure

At the meeting on January 25, the Division Commander, BG Sobke, proposed a management structure to involve the affected agencies and organizations in the LTMS development. This management structure was refined at an ensuing meeting, called on February 27, and was adopted by the group at a meeting on March 28. The adopting participants, who were designated the Policy Review Committee under the new structure, represented 32 public and private organizations. The Committee's contributions of coordinated review, compromise, and consensus-building on policy issues for the LTMS are at the heart of the management structure. The management structure also provides for an Executive Committee, a Management Committee, a Work Group and Sub-Work Groups, a Technical Advisory Panel, and public involvement through the San Francisco Estuary Project. A brief discussion of the makeup and functions of these committees is given below.

Executive Committee

This committee consists of the regional leaders of the regulatory agencies: USAE Division, South Pacific, Commander; EPA Regional Administrator; San Francisco Bay Regional Water Quality Control Board (SFBRWQCB) Chairperson; Bay Conservation and Development Commission (BCDC) Chairperson; and a State Dredging Coordinator. This committee provides policy guidance and direction on the overall conduct of the program and resolves policy issues as they arise. Their decisions, guidance, and directions are based on the approved LTMS Study Plan, which was cooperatively developed by the named agencies and the members of the Policy Review Committee as represented on the staff-level work groups.

Policy Review Committee (PRC)

This committee meets periodically to provide input to, and receive updates on, the LTMS program. The PRC provides an important public involvement and review conduit for the Executive Committee to guide the development and implementation of the LTMS. Recommendations to the Executive Committee are made by group consensus after open discussions occurring at PRC meetings. The frequency of such meetings is about every 3 months unless matters arise that dictate more frequent sessions. Where additional information or clarification is needed, the Executive Committee refers the matter to the Management Committee (described below) to provide the required data. The Executive Committee provides the PRC with periodic updates on study progress and refers other matters to them regarding the conduct of the program for comments and recommendations.

Management Committee

This committee consists of the USAE District, San Francisco, Commander; USAED Division, South Pacific, LTMS Program Manager; EPA Water Management Division Chief; SFBRWQCB Executive Officer; and BCDC Executive Officer. The committee provides the necessary interagency coordination and direction as well as facilitating in cost-participation matters. Each member of the Management Committee represents the interests of all departments and divisions of his agency, board, or commission. Each member is authorized to

speak for these entities and to enlist their resources as appropriate, and is responsible for all coordination within his respective organization. Matters that cannot be resolved by the Management Committee are referred to the Executive Committee for expeditious resolution. The Management Committee develops and disseminates study reports for review and comment to the Technical Advisory Panel and the San Francisco Estuary Project. The Management Committee also selects and appoints the outside technical experts that serve on the Technical/ Science Advisory Panel. The Management Committee exercises direct supervision over the LTMS Work Group and Sub-Work Groups.

Work Group and Sub-Work Groups

This committee consists of the USAE District, San Francisco, Commander; LTMS Project Manager; and the Sub-Work Group chairpersons (representatives of EPA (Ocean), SFBRWQCB (In-bay) and BCDC (Nonaquatic/Reuse). The responsibility for developing all of the LTMS concepts, work plans, and study reports resides in the Work Group. This responsibility extends to acquiring and allocating the necessary resources (manpower and funds) to perform the required fieldwork and to produce the study reports. Under the purview of the Work Group are three Sub-Work Groups handling the individual study elements (Ocean, In-bay, and Nonaquatic/Reuse. Sub-Work Groups consist of technical staff members from agencies and interest organizations that have subject matter knowledge or expertise in each of these areas. The entire study effort is being integrated by the Corps.

Technical Advisory Panel

The management structure includes technical review by both an expert panel and local agency/private scientists specializing in disciplines germane to the three areas of study: ocean, in-bay, and nonaquatic/reuse disposal options. The purpose of the expert panel is to provide the program with critical outside technical review of the program's conceptual approach, scientific rigor, and application of technical findings. The expert panel provides their independent, expert opinions to the Management Committee on the scientific credibility and defensibility of the work being reviewed. The Technical Advisory Panel also includes interested and knowledgeable scientists from Federal, state, and local interest organizations. The member scientists, having local subject matter expertise, evaluate study concepts and documents and provide their advice and local perspectives to the expert panel members.

San Francisco Estuary Project (SFEP)

The SFEP provides a vehicle to disseminate information concerning the LTMS to the general public through its infrastructure and public involvement/outreach programs. LTMS progress reports are provided at SFEP committee meetings and other activities and are presented through quarterly newsletters, public workshops, and fact sheets.

LTMS Process

Conceptually, LTMS development is a sequential process consisting of five phases, each with a series of essential activities that lead to decision-making before continuing to the next phase (Francingues and Mathis 1989). A description of the five LTMS phases follows.

Phase I - Evaluate Existing Management Options

This phase is intended to serve as the first level of appraisal and decision-making. The first phase of the San Francisco Bay regional LTMS, Phase I Needs Assessment, was completed by the Management Committee in October 1990, adopted by the Policy Review Committee in November 1990, and approved in December 1990 by the Executive Committee. The document concluded that there clearly is a shortfall in disposal capacity for the improvement projects scheduled by the Corps, the Navy, and the ports for this region--on the order of 19 mcy. It stated that current in-bay aquatic capacity appears to be capable of handling the maintenance dredging requirements for the foreseeable future, provided the material is processed before disposal to optimize dispersion from the site. However, there is a need to address environmental concerns. In addition, the Phase I document found that there is currently no identified aquatic or nonaquatic disposal capacity for contaminated dredged material. It emphasized that beneficial uses should be considered as a high priority for any material that meets the economic, engineering, and environmental criteria for a given use.

In summary, the Phase I document stated that the projects in the region will likely require continued use of existing disposal alternatives as well as additional open-water disposal, confined disposal, and beneficial uses to satisfy their long-term dredging requirements. At this point in the LTMS process, a decision can be made as to whether there is a need to formulate management alternatives (Phase II) or to document the long-term practicality of the existing management strategy (Phase III) prior to proceeding to implementation. In the San Francisco case, the findings of Phase I led to development of Phase II of the LTMS process.

Phase II - Formulate Alternatives

The objective of Phase II is to systematically formulate, develop, and retain all viable long-term dredged material disposal management options. It is essential that equal consideration be given to all categories of management options (upland, wetland, intertidal, open water, and structural methods to reduce dredging). The need for specific environmental, engineering, and economic studies has been determined at the local level for studies of the ocean, in-bay, and nonaquatic/reuse alternatives in the San Francisco Bay region. A Study Plan was initially developed in February 1990 and revised in April, August, and December 1990 based on comments from the Policy Review Committee, the Sub-Work Groups, and Corps Headquarters.

In developing the Study Plan, maximum use was made of existing studies and technical fieldwork directly germane to studies of the proposed alternatives. Member organizations of the Sub-Work Groups and/or the Policy Review Committee made every effort to make such studies or other information available to the Management Committee so that the previous research and findings could be effectively incorporated into the LTMS effort. The Management Committee finalized the Study Plan's description of tasks and the program budget, including the staff costs of the participating agencies, in January 1991. The documents were reviewed by the Policy Review Committee in March, prior to approval by the Executive Committee in June 1991. Once the required data requirements have been met in late 1993, the next step will be to array the alternatives and eliminate the impracticable alternatives. Finally, this array will be transformed from viable management options into attainable and implementable alternatives during Phase III.

Phase III - Detailed Analysis of Alternatives

This phase provides for a thorough analysis of existing dredged material disposal management plans and the detailed evaluation, screening, and selection of a preferred long-term disposal management strategy. The San Francisco Phase III activities are scheduled to begin in early 1994. The analyses will weigh and balance the engineering, economic, and environmental factors and benefits. The purpose of the analyses will be to select the most practicable strategy (consisting of an array of alternatives for implementation) and to provide the documentation needed to support this selection.

Phase IV - LTMS Implementation

The purpose of Phase IV is to develop the operations plan for implementing the selected dredged material disposal sites. This activity is scheduled for late summer 1994. In developing this implementation plan, considerations will include the administrative, procedural, management, and monitoring requirements.

Phase V - Periodic Review and Update

The final phase of LTMS development is the periodic reevaluation of the LTMS plan, based on changing regulatory, economic, and environmental conditions and on technologic conditions. The intent of Phase V is to ensure that decision-makers will maintain a viable implementation strategy that reflects the changing conditions. It is anticipated that there will be a biannual review of the San Francisco LTMS.

Regional Activities

Ocean studies

To meet the scheduled completion date of mid-1994 for Phase III, some of the field studies for the regional program were begun in 1990. Ocean field studies necessary to designate a new offshore disposal site under Section 102 of the Marine Protection, Research and Sanctuaries Act were initiated in July. A preliminary oceanographic survey of bathymetry in the Gulf of the Farallons was completed by EPA in August 1990. From the preliminary survey, a detailed mosaic of the ocean floor was constructed in water depths from 100 to 1,000 fathoms. This work was done by the U.S. Geological Survey and provided details on the area's bathymetry, material composition, and stability. The results of the survey allowed preparation of a detailed ocean site study plan including current measurements, numerical modeling, and benthos assessment. Current meters were deployed in the Gulf of the Farallons, and one hydrographic cruise was completed by March 1991.

A critical element in the ocean studies process is the adequate description of changes in oceanographic seasons offshore of San Francisco. This requirement for a complete record of seasonal changes demands more than a year of field investigations. The upwelling period begins in the April-May time frame and is key to the movement of nutrients and fish off the coast. To record this event, fieldwork was initiated prior to adoption of the final Study Plan.

Currently, EPA is completing its year-long ocean site characterization process, which included gathering data on sediments, currents, benthic organisms, fish, birds, mammals, and other significant study parameters at five areas offshore of San Francisco. One oceanographic survey obtained sediment profile images at 107 stations throughout the LTMS study region. EPA used these data to delineate composite areas for sediment chemistry analysis, to map sediment characteristics, and to tailor 55 box coring station locations. The field sampling collection was complemented with camera sled and remote-operated vehicle surveys to provide reliable coverage of biota at several locations.

In March 1992, the last of the LTMS oceanographic work will be completed when EPA retrieves the current meters, finishes collection of hydrographic data, and conducts its final trawl. In the meantime, analysis of previously collected sediment, organism, and photo-quadrat data is under way for inclusion in the site designation Environmental Impact Statement.

In-bay studies

The LTMS Study Plan calls for the determination of the fate and transport of in-bay disposed sediments (in both the short and long term) and assessment of the biological effects of discharged sediments. The plan proposes that this be achieved by developing an understanding of the physical system and modeling that understanding with a numerical tool. Findings would be used in future decisions as to the acceptability of in-bay disposal on biological resources and project maintenance activities. In the short term, biological effects stem principally from burial and avoidance behavior of fishes from suspended solids. The long-term effects are attributed both to potential contaminant movement from sediments into organisms (i.e., bioaccumulation) and to sediment recirculation back into project sites. In-bay studies began in fall 1991 with investigations of modeling and monitoring needs for dredged material disposal within the estuary. Activities are continuing with development of a data management structure and methods manual for chemical and biological analyses of San Francisco Bay sediments. Laboratory and field work are anticipated to begin in early summer 1992 to investigate the acceptability of present in-bay disposal practices based on a detailed plan of study currently being drafted. If existing practices and/or new techniques are determined to be environmentally acceptable, additional in-bay disposal sites may be sought during 1993.

Nonaquatic/reuse studies

The objectives of the nonaquatic/reuse disposal studies of the LTMS are to (a) identify, develop, and analyze opportunities and constraints for the use of dredged material as a resource in the San Francisco Bay and Sacramento/San Joaquin Delta areas and for the disposal of contaminated sediments; (b) analyze and resolve reuse and nonaquatic disposal constraints through working with interested parties, filling information gaps, performing demonstration projects, and recommending site-specific plans for reuse and nonaquatic disposal opportunities; and (c) develop and evaluate implementation strategies for nonaquatic disposal plans. The first of three work elements was initiated in January 1992. Each element will be conducted as a separate, but linked, study to meet the above objectives. The overall intent of the first element is to focus the efforts of the succeeding work elements on the most promising reuse opportunities and feasible contaminated sediment disposal methodologies paired with specific local upland sites.

Conclusion

A regional Long-Term Management Strategy for dredged material disposal from San Francisco Bay was established in early 1990. Involvement of multiple participants was necessary because of the conflict and confusion surrounding the issue of dredged material disposal in the region. Through the LTMS process, regional consensus is being forged. The LTMS management structure was organized in a spirit of cooperation between the major Bay Area regulatory agencies (Corps, EPA, SFBRWQCB, and BCDC) and the State of California. Conduct of the LTMS program is utilizing consensus between the involved parties at the executive, management, and working levels. This program, which holds such enormous consequences for the future well-being of the region's economy and environment, is proceeding on schedule with excellent participation from regional navigation interests.

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Dioxin Toxic Equivalents (TEQs) in Dredged Sediment Evaluation

by
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Introduction

Polychlorinated dibenzo-*p*-dioxins (PCDDs), especially 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD), are among the most toxic and persistent of environmental contaminants. These and the structurally similar polychlorinated dibenzofurans (PCDFs), polychlorinated biphenyls (PCBs), and other groups of polyhalogenated aromatic hydrocarbons (PHHs) are associated with genotoxic and cytotoxic effects, as well as body weight loss, reproductive impairment, acute lethality, chloracne, liver damage, edema, and other toxicities (Kociba et al. 1978, Greig 1979, Kociba and Cabey 1985, Safe 1987). Much concern has arisen in recent years over the widespread occurrence and potential for toxicity of these chemicals in the aquatic environment, including sediments slated for dredging and disposal.

Most dioxin research to date has focused on 2,3,7,8-TCDD. Nevertheless, there are thousands of other PHH compounds, including 75 PCDD congeners and 135 PCDF congeners, and it is appealing to try to understand the potential toxicity of some of these related compounds in terms of the more familiar (and most toxic) 2,3,7,8-TCDD. Thus, dioxin "toxic equivalents" (TEQs) have been formulated in an attempt to express the combined toxicity of a mixture of PHH in a sample as though the sample contained an equivalent amount of 2,3,7,8-TCDD alone. The rationale for TEQs is the strong correlation between structural similarity of PHH congeners to 2,3,7,8-TCDD and toxicity. PHH congeners most similar in size and shape to the 2,3,7,8-TCDD molecule have the highest binding affinity for the cellular *Ah* receptor protein,[4] and the highest toxicity, relative to 2,3,7,8-TCDD. The toxicity of a PHH thus can be expressed as a fraction of 2,3,7,8-TCDD toxicity dependent on the degree of structural similarity.

This paper describes the use of TEQs in the regulatory decision-making process involving dioxin-containing dredged sediments. Shortcomings in the present use of TEQ methodology are outlined and supported by examination of recent cases where TEQs have been used in

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[2] AScI Corporation, Vicksburg, MS.

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[4] For a full discussion of the nature of the *Ah* receptor and its role in PHH toxicity, see McFarland et al. (in review).

regulatory decisions without regard to bioavailability of TCDD-like compounds from sediments. An alternative approach based on bioassay-derived TEQs shows promise in overcoming many of the problems associated with TEQs as currently derived from chemical analysis.

Shortcomings of TEQs in Environmental Assessments

Dioxin TEQs were standardized in 1988 using International Toxicity Equivalency Factors (I-TEFs) (Table 1). The concentrations of the listed PHH congeners in a sample are multiplied by their respective I-TEFs, and the products are summed to obtain a TEQ. The derivation of I-TEFs was based on several criteria, the most important of which were long-term carcinogenicity studies (Committee on the Challenges of Modern Society 1988, Kutz et al. 1990, Safe 1990). As such, I-TEFs ignore the large (several orders of magnitude) species- and response-dependent variability in toxic equivalent factors. All data used in the derivation of I-TEFs were obtained from mammalian (primarily rodent) studies. I-TEFs were intended for the protection of human health against the risk of cancer. Thus, the relevance of their application in aquatic environment impact assessments is questionable.

Table 1
International Toxicity Equivalency Factors (I-TEF)

PCDD Congener	I-TEF	PCDF Congener	I-TEF
2,3,7,8-TCDD	1	2,3,7,8-TCDF	0.1
1,2,3,7,8-PeCDD	0.5	2,3,4,7,8-PeCDF	0.5
		1,2,3,7,8-PeCDF	0.05
1,2,3,4,7,8-HxCDD	0.1	1,2,3,4,7,8-HxCDF	0.1
1,2,3,7,8,9-HxCDD	0.1	1,2,3,7,8,9-HxCDF	0.1
1,2,3,6,7,8-HxCDD	0.1	1,2,3,6,7,8-HxCDF	0.1
		2,3,4,6,7,8-HxCDF	0.1
1,2,3,4,6,7,8-HpCDD	0.01	1,2,3,4,6,7,8-HpCDF	0.01
		1,2,3,4,7,8,9-HpCDF	0.01
OCDD	0.001	OCDF	0.001

I-TEFs have been formulated for 17 PCDD and PCDF congeners containing the chlorine 2,3,7,8-substitution pattern. Not included are the PCBs and other structurally related PHHs. Some of these compounds, particularly the coplanar PCBs, are frequently much more abundant in the environment and thus may pose a greater toxic threat to both wildlife and humans than do the dioxins and furans (Tanabe et al. 1987a,b; Niimi and Oliver 1989; Dewailly et al. 1991). Safe (1990) proposed an expansion of the I-TEF list to include coplanar polychlorinated and polybrominated biphenyls, along with brominated and bromo/chloro dibenzo-*p*-dioxins and dibenzofurans. Inclusion of the several hundred compounds in these categories might increase the protectiveness of TEQs somewhat in environmental applications, but would

also drive the already highly expensive chemical analytical cost ($1,000 to $2,000 per sample for the 17 PCDDs and PCDFs) to astronomical levels.

Because I-TEFs are summed to obtain a TEQ, additivity of toxic effect of the individual PCDD and PCDF congeners is assumed, and possible synergism or antagonism is ignored. In fact, antagonistic effects among PHH congeners in a mixture have been demonstrated in a number of cases. The PCB mixture Aroclor 1254, other Aroclor mixtures, and specific individual PCB, PCDD, and PCDF congeners have all been shown to antagonize the effects of 2,3,7,8-TCDD (Bannister et al. 1987; Haake et al. 1987; Astroff, Romkes, and Safe 1989; Waern et al. 1989, 1990; Davis and Safe 1990; Prokipcak et al. 1990).

The current method of calculating TEQs from I-TEFs and analytical chemistry thus has many shortcomings, not the least of which is high cost, which make the method unsuitable for environmental regulatory evaluations. In fact, the I-TEF method was intended by its developers to be only an interim approach that should be replaced, as soon as practicable, by a more definitive bioassay for the determination of TEQs (Kutz et al. 1990, Barnes 1991).

Dredged Sediment Evaluations Using TEQs

Despite their shortcomings, TEQs have recently been required in some environmental assessments. The State of Oregon, for example, has promulgated recommendations on the use of TEQs in environmental regulations (Oregon Department of Environmental Quality (DEQ) 1990). The U.S. Environmental Protection Agency (USEPA) has adopted TEQs in risk assessment and in rule making, but has not been consistent in its application. For example, in a recent regulatory decision, Region X of the USEPA, in conjunction with the Oregon DEQ, the Washington Department of Ecology, and the Idaho Department of Environmental Quality, set a total maximum daily loading value of 6 mg/day 2,3,7,8-TCDD for the Columbia River Basin based solely on water quality criteria for 2,3,7,8-TCDD, not on TEQs.

On the other hand, several U.S. Army Corps of Engineers (USACE) elements have recently been asked to use TEQs rather than actual concentrations of 2,3,7,8-TCDD in decision making for Federal navigation projects. One such case involved a risk assessment performed by the U.S. Army Engineer District (USAED), Seattle, in conjunction with maintenance dredging of the Federal channel at Gray's Harbor, Washington (USACE 1991). Several tiers of the dredged sediment evaluation tiered testing protocol outlined in the "Green Book" (USEPA/USACE 1991) were performed concurrently to save time. 2,3,7,8-TCDD was detected in only 3 of 17 sediments, at concentrations ranging from 1.5 to 3.9 parts per trillion (pptr). 2,3,7,8-substituted PCDDs were present in some sediment samples, but at such low concentrations that there was no "reason to believe" in a Tier II evaluation of the sediments that dioxin would be bioaccumulated to detectable levels. All sediment toxicity tests were negative, and bioaccumulation tests were inconclusive; thus, there were no Tier III exceedances. Nonetheless, the District was compelled by USEPA and state agencies to perform a TEQ-based human health risk analysis on the project sediments. The risk analysis was performed with data generated by assuming concentrations to be equal to one-half the detection limit since most samples contained no detectable dioxins or furans. The outcome of the risk assessment was no incremental human health risk attributable to these compounds.

In another case, the USAED, Walla Walla, was delayed in 1991 from performing a previously approved maintenance dredging project in the upper Snake River when the "104 Mill Survey" identified a nearby industrial source of dioxin. This delay was resolved by an

agreement between the District and USEPA Region X to sample sediments slated for dredging for selected dioxin and furan congeners. Because the cost of dioxin determinations is so high, the District proposed a plan whereby dioxin would be analyzed only in sediments with the highest total organic carbon (TOC) content (where dioxin could be expected to be found if present). Sediments were collected throughout the project area, and TOC was determined in all samples. The sediment samples were archived until initial dioxin testing of the highest-TOC samples was complete. If dioxins were found in the high-TOC samples, the next highest-TOC samples would be analyzed. The analytical results would be used to calculate TEQs. This evaluation is in progress, and results have not yet been published.

In a third case involving TEQs, the National Oceanographic and Atmospheric Administration Natural Resources Trustees recently presented the USAED, Charleston, with concerns regarding dioxin contamination in Winyah Bay, South Carolina. As a result of the "104 Mill Survey," the South Carolina Department of Health and Environmental Control (SCDHEC) sampled organisms and sediments throughout Winyah Bay. They found a few organisms with elevated levels of dioxin TEQs, and 5 of 11 sediment samples had dioxin TEQ levels above 2 pptr. In January and February 1989, 22 stations were sampled for organisms. Of these samples, fourteen exceeded 1 pptr TEQs, and three had TEQs exceeding the 25-pptr U.S. Food and Drug Administration (FDA) limit for 2,3,7,8-TCDD in edible fish portions. In August and September 1989, SCDHEC sampled 51 organisms for dioxins. Of these, twenty-four had TEQs exceeding the 1-pptr detection limit routinely obtained for dioxin in tissue samples, and one exceeded the 25-pptr FDA limit.[1] Congeners analyzed in the tissue samples were the 17 listed in Table 1; of these, the most frequently occurring were 2,3,7,8-TCDD, OCDD, and 2,3,7,8-TCDF. The Charleston District will be evaluating Federal project sediments for three reaches of Winyah Bay using guidance published in the "Green Book" (USEPA/USACE 1991).

Regulatory evaluations of dioxin-containing sediments in the New York-New Jersey Harbor area have been based on bioaccumulation of 2,3,7,8-TCDD, rather than on TEQs. Bioaccumulation testing using the polychaete *Nereis virens* is performed if dredging project sediments exceed 1 pptr 2,3,7,8-TCDD. In 1992, the USAED, New York, proposed guidelines for evaluating dioxin bioaccumulation data.[2] If bioaccumulation levels in worms exposed to the dredged sediment were significantly greater (95 percent confidence level) than bioaccumulation levels in worms exposed to reference sediment, the following would apply. For bioaccumulation of at least 1 pptr 2,3,7,8-TCDD and less than 10 pptr in worms exposed to the dredged sediment, ocean disposal would be allowed and expeditious capping required (within 2 weeks, 2 to 1 ratio of cap to capped material). For bioaccumulation of at least 10 pptr and less than 25 pptr, expeditious capping would be required (within 10 days, at least 2 to 1 ratio of cap to capped material), and special measures (e.g., onboard inspectors) would be taken to ensure that the material was accurately placed and capped. For bioaccumulation of 25 pptr and above, ocean disposal would not be allowed. These protocols have been accepted by the USEPA Region II and are to be reassessed within 1 year to 18 months after completion of the first dredging project involving dioxin evaluation.

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[1] SCDHEC, unpublished data.

[2] Letter, 11 Mar 1992, from John F. Tavolaro, USAED, New York, to members of Federal Interagency Dioxin Steering Committee regarding decision-making framework for ocean disposal of dredged material containing 2,3,7,8-TCDD.

As the above examples demonstrate, the regulation of dioxin-containing sediments is far from standardized on a national basis. More research into the relationship between sediment levels and toxicity is certainly required.

Bioaccumulation and Bioavailability of Dioxins and Furans from Sediments

The toxic effects of dioxins in animals have been demonstrated primarily through laboratory studies using rodents (Kociba et al. 1978, Kociba and Cabey 1985), as well as assays involving aquatic organisms (Cooper 1989, van der Weiden et al. 1989, Wisk and Cooper 1990) and bird embryos (Hoffman et al. 1987; Tillet, Ankley, and Geisy 1989). Dioxins and related compounds have been implicated in the poor reproductive success observed in Forster's terns from an area of high contamination in Green Bay, Wisconsin (Hoffman et al. 1987). However, there has been little evidence to date directly linking dioxins in sediments with toxic effects in the aquatic environment. Bioavailability of these compounds in sediments appears to be quite limited, and as mentioned earlier, the relatively much more abundant PCBs may have greater toxic potential than dioxins and furans.

Studies are few in which dioxins and furan residues have been analyzed in field-collected aquatic organisms as well as sediments from the corresponding location (Reilly et al., in preparation). Table 2 lists concentrations of PCDD and PCDF congeners in sediments and tissues of fish collected from locations in the Great Lakes region. In two of the three studies, only 2,3,7,8-TCDD or 2,3,7,8-TCDD and 2,3,7,8-TCDF were measured. Congener residues in fish seldom exceeded those of the corresponding sediment, ranging from undetected to 60 picograms per gram (pg/g; = pptr). Gardner and White (1990) noted that reported levels of PCDF congeners in fishes from various estuaries, lakes, and rivers rarely exceeded 100 pg/g.

Kuehl et al. (1987) analyzed a large number of PCDD and PCDF congeners in sediment and carp collected from Petenwell Reservoir, Wisconsin River, Wisconsin (Table 3). In only one case did a congener concentration (2,3,4,6,7-pentaCDF) in fish exceed that of the sediment, and then by less than 1 pg/g. Most of the PCDD and PCDF congeners found in the sediment, including some occurring in relatively high concentrations, were undetected in the fish. In general, the 2,3,7,8-substituted congeners were preferentially bioaccumulated. We calculated the theoretical bioaccumulation potential (TBP) for the PCDD and PCDF congeners based on their sediment concentrations and the stated sediment organic carbon content of 3.1 percent and organism lipid content of 8 percent (Kuehl et al. 1987), using the formula in the "Green Book" (USEPA/USACE 1991, p 10-7). For a neutral organic chemical, TBP expresses the theoretical maximum bioaccumulation from sediment that can be expected in an organism of given lipid content. TBP assumes chemical equilibrium partitioning among the phases of a system, yet equilibrium is seldom, if ever, achieved in natural aquatic environments (Clarke, McFarland, and Dorkin 1988).

Calculated TBPs were an order of magnitude higher than the sediment concentrations of the PCDDs and PCDFs (Table 3). The amounts actually bioaccumulated by the fish were far less than predicted by their TBPs. Dividing actual bioaccumulation by TBP provides a measure of bioavailability, p (Clarke and McFarland 1991). A p value of 1 indicates completed bioavailability of a neutral organic chemical from sediment. Values of p greater than 1 would suggest bioaccumulation from other sources, e.g. food, in addition to sediment. Values of p for the PCDD and PCDF congeners were extremely low (Table 3), ranging from less than

Table 2
Dioxin and Furan Concentrations in Sediments and Field-Collected Fish from Selected Locations

Congener	Sediment Concentration pg/g	Tissue Concentration pg/g	Fish Species
	Lake Ontario, Station 208 (Short 1989)		
2,3,7,8-TCDD	38[1]	12	Slimy sculpin
	Lake Ontario, Station 210 (Short 1989)		
2,3,7,8-TCDD	63.7[1]	41	Slimy sculpin
	Green Bay, Lake Michigan (Smith et al. 1990)		
2,3,7,8-TCDD	1	< 1	Fathead minnow
2,3,7,8-TCDF	28	7	Fathead minnow
	Fox River, Wisconsin (Smith et al. 1990)		
2,3,7,8-TCDD	14	0.8	Spottail shiner
		1	Bullhead
		5	Walleye
2,3,7,8-TCDF	84	60	Spottail shiner
		5	Bullhead
		47	Walleye
	Menominee River, Wisconsin (Smith et al. 1990)		
2,3,7,8-TCDD	< 2	1	Bullhead
2,3,7,8-TCDF	27	20	Bullhead
	Lake Orono, Minnesota (Reed et al. 1990)		
2,3,7,8-TCDD	ND, ND, ND[2]	ND, ND, ND	Carp
1,2,3,7,8-PeCDD	ND, ND, ND	ND, ND, ND	Carp
1,2,3,4,7,8-HxCDD	ND, ND, ND	ND, ND, ND	Carp
1,2,3,6,7,8-HxCDD	ND, ND, ND	ND, 2.7, ND	Carp

(Continued)

[1] 0- to 3-cm sediment sample.
[2] ND = not detected; detection limits: 0.61 to 4.1 pg/g for sediments, 0.28 to 6.6 pg/g for fish.

Table 2 (Continued)

Congener	Sediment Concentration pg/g	Tissue Concentration pg/g	Fish Species
1,2,3,7,8,9-HxCDD	ND, ND, ND	ND, ND, ND	Carp
1,2,3,4,6,7,8-HpCDD	52, 56, ND	ND, 11, 10	Carp
OCDD	530, 600, 490	39, 35, 43	Carp
2,3,7,8-TCDF	ND, 0.31, ND	1.3, 1, 1.1	Carp
1,2,3,7,8-PeCDF	ND, ND, ND	ND, ND, ND	Carp
2,3,4,7,8-PeCDF	ND, ND, ND	ND, 2, ND	Carp
1,2,3,4,7,8-HxCDF	ND, ND, ND	ND, ND, ND	Carp
1,2,3,6,7,8-HxCDF	ND, ND, ND	ND, ND, ND	Carp
1,2,3,7,8,9-HxCDF	ND, ND, ND	ND, ND, ND	Carp
2,3,4,6,7,8-HxCDF	ND, ND, ND	ND, ND, ND	Carp
1,2,3,4,6,7,8-HpCDF	11, 6.9, ND	ND, ND, ND	Carp
1,2,3,4,7,8,9-HpCDF	ND, ND, ND	ND, ND, ND	Carp
OCDF	ND, ND, ND	ND, ND, ND	Carp

Table 3
Sediment and Tissue (Carp) Concentrations, Theoretical Bioaccumulation Potential (TBP), and Bioavailability of Dioxins and Furans from Petenwell Reservoir, Wisconsin River, Wisconsin (Kuehl et al. 1987)

Congener	Concentration, pg/g			Bioavailability
	Sediment	Carp	TBP[1]	*p*
1,3,6,8-TCDD	17	ND[2]	175	<0.01
2,3,7,8-TCDD[3]	**170**	**120**	**1,755**	**0.068**
1,2,4,6,8-; 1,2,4,7,9-PCDD	136	ND	1,404	<0.01
1,2,3,6,8-PeCDD	53	ND	547	<0.01
1,2,4,7,8-PeCDD	36	ND	372	<0.01
1,2,3,7,9-PeCDD	15	ND	155	<0.01
1,2,3,4,7-; 1,2,4,6,9-PeCDD	53	ND	547	<0.01
1,2,3,7,8-PeCDD	**31**	**4.8**	**320**	**0.015**
1,2,3,6,9-PeCDD	14	ND	145	<0.01
1,2,4,6,7-; 1,2,4,8,9-PeCDD	23	ND	237	<0.01
1,2,3,6,7-PeCDD	11	ND	114	<0.01
1,2,3,8,9-PeCDD	5	ND	52	<0.02
1,2,4,7,9-PeCDD; 1,2,4,6,8,9-; 1,2,3,4,6,8-HxCDD	1,090	ND	11,252	<0.01
1,2,3,6,7,9-; 1,2,3,6,8,9-HxCDD	580	ND	5,987	<0.01
1,2,3,6,7,8-HxCDD	**180**	**16**	**1,858**	**0.009**
1,2,3,4,6,9-HxCDD	16	ND	165	<0.01
1,2,3,7,8,9-HxCDD	**60**	**ND**	**619**	**<0.01**
1,2,3,4,6,7,8-HpCDD	**2,190**	**27**	**22,606**	**0.001**
1,2,3,4,6,7,9-HpCDD	4,720	ND	48,723	<0.01
OCDD	**20,560**	**25**	**212,232**	**0.0001**
1,3,7,8-TCDF	6	ND	62	<0.02
1,3,4,6-; 1,2,4,8-TCDF	20	ND	206	<0.01
1,2,4,6-TCDF	14	ND	145	<0.01
1,2,3,7-; 1,2,6,8-; 1,4,7,8-; 1,3,6,9-TCDF	15	ND	155	<0.01
1,2,3,8-; 1,4,6,7-; 2,4,6,8-; 1,2,3,6-TCDF	31	ND	320	<0.01
1,2,7,8-TCDF	88	ND	908	<0.01
1,2,6,7-; 1,2,7,9-TCDF	10	ND	103	<0.01
1,2,4,9-; 2,3,6,8-TCDF	19	ND	196	<0.01

(Continued)

[1] Based on 8 percent organism lipid content and 3.1 percent sediment organic carbon.
[2] Not detected; minimum level of detection, 1 pg/g.
[3] 2,3,7,8-substituted congeners in bold.

Table 3 (Continued)

Congener	Concentration, pg/g			Bioavailability
	Sediment	Carp	TBP[1]	p
2,4,6,7-TCDF	7	ND	72	<0.02
2,3,7,8-TCDF	**182**	**28**	**1,879**	**0.015**
2,3,6,7-TCDF	24	ND	248	<0.01
3,4,6,7-TCDF	5	ND	52	<0.02
1,2,8,9-TCDF	8	ND	83	<0.02
1,2,4,6,8-PeCDF	64	ND	661	<0.01
1,2,3,6,8-; 1,3,4,7,9-PeCDF	9	ND	93	<0.02
1,2,4,7,8-PeCDF	22	ND	227	<0.01
1,2,4,7,9-; 1,3,4,6,7-PeCDF	3	ND	31	<0.04
1,2,4,6,7-PeCDF	8	ND	83	<0.02
1,2,3,4,7-; 2,3,4,6,9-PeCDF	4	ND	41	<0.03
1,2,3,4,8-; **1,2,3,7,8-PeCDF**	**14**	**2.6**	**145**	**0.018**
2,3,4,6,8-; 1,2,4,6,9-PeCDF	9	ND	93	<0.02
2,3,4,8,9-PeCDF	6	ND	62	<0.02
1,2,4,8,9-PeCDF	5	ND	52	<0.02
2,3,4,7,8-PeCDF	**8**	**4.4**	**83**	**0.053**
1,2,3,8,9-PeCDF	2	ND	21	<0.05
2,3,4,6,7-PeCDF	2	2.8	21	0.133
1,2,3,4,6,8-HxCDF	21	ND	217	<0.01
1,3,4,6,7,8-HxCDF	91	ND	939	<0.01
1,2,3,4,7,9; **1,2,3,4,7,8-HxCDF**	**30**	**ND**	**310**	**<0.01**
1,2,3,6,7,8-HxCDF	**11**	**1**	**114**	**0.009**
1,2,3,4,6,7-HxCDF	84	2	867	0.002
1,2,3,6,8,9-; 1,2,3,4,8,9-HxCDF	6	ND	62	<0.02
1,2,3,4,6,7,8-HpCDF	**290**	**2.5**	**2,994**	**0.0008**
1,2,3,4,6,8,9-HpCDF	430	ND	4,439	<0.01
OCDF	**850**	**ND[1]**	**8,774**	**<0.01**

[1] Detected but did not meet quality assurance criterion at 5 pg/g.

0.01 for most congeners to 0.068 for 2,3,7,8-TCDD and 0.133 for 2,3,4,6,7-pentaCDF. These low values suggest that partitioning of PCDDs and PCDFs between sediments and fish had not reached equilibrium. Highly hydrophobic chemicals such as the PCDDs and PCDFs may take long periods (more than a year) to completely desorb from sediment; thus, their bioavailability from sediment would be extremely limited (Clark and McFarland 1991).

Because bioavailability of PCDDs and PCDFs from sediment is so low, calculation of TEQs from sediment concentrations of these congeners, or especially from predicted bioaccumulation measures such as TBP, could greatly overstate the potential toxicity to aquatic organisms of dioxins and furans in sediment. Table 4 shows the TEQs calculated from Petenwell Reservoir sediment, fish, and TBP concentrations of the PCDD and PCDF congeners for which I-TEFs have been determined. The TEQ calculated from actual bioaccumulation was less than half that calculated from sediment concentrations, and less than 1/20 the TEQ calculated from TBP.

Table 4
Dioxin Toxic Equivalents (TEQs) Calculated from Petenwell Reservoir Sediment, Carp, and TBP Concentrations of Toxicologically Active Dioxin and Furan Congeners

		I-TEF × Concentration, pg/g		
Congener	I-TEF	Sediment	Fish	TBP
2,3,7,8-TCDD	1	170	120	1,755
1,2,3,7,8-PeCDD	0.5	15.5	2.4	160
1,2,3,6,7,8-HxCDD	0.1	18	1.6	186
1,2,3,7,8,9-HxCDD	0.1	6	--	61.9
1,2,3,4,6,7,8-HpCDD	0.01	21.9	0.27	226
OCDD	0.001	20.6	0.025	212
2,3,7,8-TCDF	0.1	18.2	0.8	188
1,2,3,7,8-PeCDF	0.05	0.7	0.13	7.25
2,3,4,7,8-PeCDF	0.5	4	2.2	41.5
1,2,3,4,7,8-HxCDF	0.1	3	--	31
1,2,3,6,7,8-HxCDF	0.1	1.1	0.1	11.4
1,2,3,4,6,7,8-HpCDF	0.01	2.9	0.025	29.9
OCDF	0.001	0.85	--	8.77
TEQ		**283**	**128**	**2,919**

Conclusions

Dioxin TEQs are beginning to play a role in environmental evaluations including regulatory decision making involving dredged sediments. Although the calculation of TEQs has been standardized using I-TEFs, their application by state and Federal regulatory agencies is by no means consistent. In some cases, TEQs have been calculated from sediment concentrations of dioxins and furans, yet the bioavailability of these contaminants to aquatic organisms from sediments appears to be quite low, and the TEQs may considerably overstate their toxic

potential. The use of TEQs in environmental evaluations has many other shortcomings, some of which are discussed in detail in McFarland et al. (in review). In brief, the current method of analytical chemistry-based TEQs

- Includes only 17 PCDDs and PCDFs and does not include PCBs or other structurally related PHHs, some of which are much more abundant in the environment and thus may have greater toxic potential than the dioxins and furans.

- Necessitates highly expensive trace chemical analysis.

- Assumes additivity of toxic effect of the congeners in a mixture, whereas antagonistic effects have been demonstrated.

- Ignores the large (several orders of magnitude) species- and response-dependent variability in toxic equivalent factors.

- Is biased toward human health protection and may not accurately assess the real toxicity of dredged material to aquatic biota.

- Was intended only to be an interim approach until the development of an integrative bioassay.

Nevertheless, TEQs represent a way to express the toxicity of complex mixtures of environmental contaminants that is highly appealing for its simplicity. Basing TEQs on an integrative bioassay rather than on trace chemical analysis would overcome all of the problems mentioned above while retaining the simplicity of a single 2,3,7,8-TCDD-equivalent number. One such bioassay is the H4IIE in vitro bioassay, which uses the rat hepatoma H4IIE cell line (Bradlaw and Casterline 1979). This bioassay integrates the additive and antagonistic effects of a mixture into a numerical result (i.e., TEQ) at a cost per sample of 10 to 20 times less than trace chemical analysis. The H4IIE bioassay is currently under development at the U.S. Army Engineer Waterways Experiment Station (WES) for application to sediment evaluations, and will be described in detail in a future WES Environmental Effects of Dredging Technical Note.

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The Corps of Engineers Ocean Disposal Database (ODD)

by
Charles H. Lutz,[1] Barbara B. Hamilton,[2] and Thomas D. Wright[1]

Introduction

The United States is a signatory to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, more commonly referred to as the London Dumping Convention (LDC). The LDC requires that each of the 61 signatory nations submit an annual report on materials disposed in the ocean. The report is to contain information on the nature and amount of material, method of dredging and disposal, location of disposal site, and other topics to characterize the dredging and disposal activity. This requirement became effective in 1975.

To provide guidance on this requirement, the Corps issued Engineer Regulation 1145-2-308 in 1978. This regulation established a common reporting format that was based on dredged material testing and evaluation procedures in effect at that time. A separate form for each Federal project or permit was submitted by the field to Headquarters, USACE. A summary was prepared, the forms duplicated, and bound copies were provided to the LDC.

In 1991, development of a PC-compatible reporting system was initiated to simplify and standardize data submission by the field for the LDC report. In addition, as the Congress has eliminated all ocean disposal activities except for dredged material, there has been increased scrutiny of dredged material by the Congress, Federal and state agencies, and special interest groups. This has resulted in numerous requests for information, many of which are highly specific or unique. Examples of such requests include summaries of disposal volumes through time for a specific disposal site, U.S. Environmental Protection Agency Region, state, or geographic region; management and monitoring activities; and characterization of the material. Without a PC-compatible system, the extraction and compilation of information from hard copy to respond to such requests in a timely manner was virtually impossible.

The first version of the ODD system was completed in early 1992, and data from 1976-1987 have been entered into the system. As some of the data may have been lost in the Pulaski Building fire, system output is considered **preliminary** until field-verified. Data from 1988-1991 will be entered as they become available.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

[2] AScI Corporation, Vicksburg, MS.

System Description

The ODD software is available on a 3.5-in. DSHD floppy disk (unless a different size is requested). The software is written in dBase IV and contains a runtime version of dBase IV. All files needed to run the program are included. The user need not be familiar with dBase IV to use the program, as it is completely menu driven and features on-line help in many places throughout the program.

The ODD software consists of two parts. The main program and database will reside at the USAE Waterways Experiment Station (WES) to allow upward reporting as data are requested. Stand-alone runtime versions will be distributed to the field for data entry and subsequent transmittal to WES.

The speed of the program is directly related to the power of the computer equipment on which it is installed. Although it will run on any machine with a hard drive and sufficient memory, it would be quite slow, for example, if run on an 8-MHz 80286 computer (such as a Zenith Z-248). It is recommended that the software be installed on the fastest available computer.

The software will need approximately 3 MB of hard disk space and 540 KB of free RAM and must be installed on a hard drive. If free RAM is not adequate, the user will receive an error message, and the program will terminate. This may require the temporary removal of some programs from memory. As networks commonly require large amounts of RAM, difficulty may be encountered with computers linked to a network, and it may be necessary to temporarily disconnect from the network.

Input Requirements

The initial entry point to the system is by calendar year. Specific projects are characterized by name, location, and whether they are Federal, permitted, new work, or maintenance. The method(s) of dredging and disposal, frequency and volume of disposal, and the disposal site are entered for each project. The ODD contains the name and coordinates of the ocean disposal sites within each Corps District, so this need not be provided. Information regarding disposal site management and monitoring is entered.

Physical

Entry of physical data is straightforward and consists of cohesiveness, percent water, total solids, and percent sand, silt, and clay.

Chemical

The entry of chemical data for whole sediment (bulk analysis) and the water column (elutriate) posed a number of problems. In particular, detection limits and analytical methods have changed over the years and also vary from laboratory to laboratory and day to day. Further, some contaminants are reported in various ways. For example, hexachlorobenzene may be reported as such or as its isomers and polychlorinated biphenyls appear as total, Aroclors, and congeners. Hence, a concerted attempt was made to include previously analyzed contaminants and potential future contaminants in the ODD.

For each contaminant, entry consists of the number of observations, the detection limit, the number of observations that exceeded the detection limit, the lowest value, the highest value, and the mean of the values above the detection limit. It is important to realize that the mean value for a contaminant **does not** represent the actual amount present in all of the project material and **cannot** be used in mass balance calculations because the volume of material represented by one observation may be very much different from that of another observation. The contaminant information will, however, provide a crude indication as to whether a particular contaminant is changing through time.

Biological

The testing and evaluation of dredged material for ocean disposal often includes bioassays. These may consist of acute water column (dissolved and/or suspended particulate phase), acute benthic (solid phase), or estimates of the bioaccumulation potential of the solid phase. In general, the results of such tests are compared to a reference (rather than a fixed number, standard, or criterion, other than Food and Drug Administration action levels), to evaluate the suitability of the material for ocean disposal. Because of the reference approach, material that is deemed suitable for one disposal site may not be suitable for another, even though the test results are the same. Therefore, reporting absolute test results is not appropriate. As it is understood that material actually disposed met the criteria for suitability, the only information to be provided to the ODD is the genus and species of the organism(s) used in bioassays.

Other

Additional information that is requested concerns material that was proposed for ocean disposal but was not disposed. The purpose for obtaining this information is to assist in determining the reason for non-ocean disposal. Reasons might include failure to meet biological requirements, water quality criteria, lack of site capacity, logistics, safety, etc. If this portion of the ODD is applicable, other information on the ultimate disposition of the material, such as 404 disposal, beneficial uses, upland, or project cancellation/postponement, is requested. This information will be used in determining trends in ocean disposal of dredged material.

System Output

It is emphasized that the ODD includes only dredged material disposal regulated by 40 CFR 220-228 and **specifically** does not include that regulated by 40 CFR 230 (although certain aspects of the latter may have to be considered; see 33 CFR 336.2(c) for guidance as to applicability).

The ODD at WES can provide a variety of numeric and graphic outputs. These include simple lists, such as volumes disposed at a particular site, within a Corps District, state, or geographic area. Concentrations of contaminants (within the limitations described above) at a particular project at a given time or through time, bioassay organisms, physical variables, or permutations of these are available. Some data, such as management, monitoring, and material considered for ocean disposal but not disposed, will be available only for post-1987 data as these were not included in the historic database.

Additional information on the ODD may be obtained from Charles H. Lutz (601-634-2489) or Thomas D. Wright (601-634-3708). Mailing address is U.S. Army Engineer Waterways Experiment Station, ATTN: CEWES-ES-R, 3909 Halls Ferry Road, Vicksburg, MS 39180-6199.

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Initial Development of a Chronic Sublethal Bioassay for the Evaluation of Dredged Material

by
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Abstract

Sediment bioassays used in the regulatory evaluation of dredged material generally measure survival of test species following acute exposure. Recent revisions of the Section 103 Implementation Manual (Marine Protection, Research, and Sanctuaries Act, Public Law (PL) 92-532) for ocean disposal permit the use of technically sound chronic sublethal tests once they are developed. Similar provisions will likely be included in revisions to the Section 404 Implementation Manual (Clean Water Act, PL 92-500). We have initiated development of a chronic sublethal sediment bioassay with the infaunal sediment-ingesting polychaete *Neanthes arenaceodentata*. Growth and reproduction are the ecologically important sublethal end points of interest. Appropriate portions of the life cycle for measuring effects on growth were first determined by characterizing worm growth from the emergent juvenile (EJ) stage to sexually mature adults. The importance of several nontreatment factors (grain size, intraspecific density, low salinity, ammonia toxicity, etc.) on the measure of growth was determined. Standardized internal quality control procedures were developed using a standard reference toxicant (cadmium chloride).

Prior to regulatory use, any sediment bioassay should be accompanied by technically sound biologically relevant interpretive guidance. To provide this initial guidance, the relationship between growth and reproduction was established. Juvenile *N. arenaceodentata* were exposed to five geometrically decreasing food rations. Growth was measured after 3, 6, and 9 weeks. Treatment effects on growth were graded and significantly different at each time interval. The magnitude of these differences increased with time, emphasizing the importance of early juvenile growth. Reproduction, measured as fecundity (egg production) and numbers of EJ produced, was followed in groups of worms from each treatment. Reduced reproduction mirrored the reduced growth rates. This relation was very linear, with regression coefficients (r^2) of 0.99 and 0.93 for fecundity and EJ production, respectively. A draft test protocol for evaluating the chronic sublethal effects of contaminated sediments is presented. Requirements for future test development activities are identified.

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Combined Hydrodynamic and Water Quality Modeling of Lower Green Bay

by
David J. Mark,[1] Barry W. Bunch,[1] and Norman W. Scheffner[1]

Introduction

As one component of its mission, the U.S. Army Engineer District, Detroit, is responsible for maintaining the navigation channel servicing the Port of Green Bay, Wisconsin. The District deposits dredged material in a confined disposal facility (CDF) named Kidney Island, located close to Green Bay's southern shore (Figure 1). Originally constructed in 1979, Kidney Island will reach its capacity within 2 years, i.e., in 1993, necessitating the development of a new CDF. One possible solution involves expanding Kidney Island. However, because the CDF is in proximity to the Fox River mouth, and based on the high wasteloads exiting the Fox River, concern exists that the expansion may adversely affect water quality in the lower bay. By modifying the current patterns, greater quantities of pollutants may be transported into areas adjacent to the CDF which serve as spawning waters.

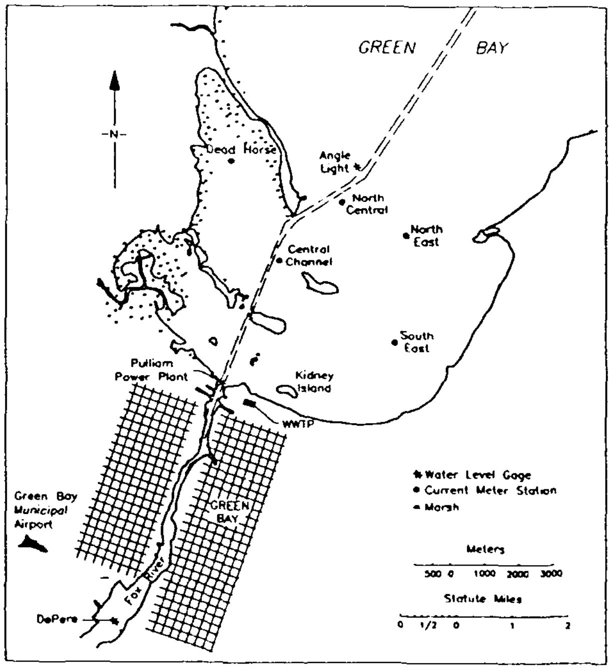

Figure 1. Locations of water surface gages and current meters

[1] U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

This paper describes the hydrodynamic and water-quality modeling approach used to determine if the proposed expansion of Kidney Island will have an adverse impact on existing water quality conditions. This assessment will be made by comparing spatial and temporal variations in dissolved oxygen (DO) concentrations between pre- and post-expansion CDF configurations. Because this is an ongoing study, this paper only discusses findings completed to date, and results are preliminary.

Site Description

Bordered by the state of Wisconsin to its south and the northern peninsula of the state of Michigan to its north, Green Bay adjoins Lake Michigan in the northwest reaches of the lake. Green Bay is the largest of Lake Michigan's bays, and its watershed accounts for approximately one third of the lake's total drainage area (Gottlieb, Saylor, and Miller 1990). The bay has an elongated and somewhat elliptical shape, with its longitudinal axis in the south-southwest to north-northeast direction. Basin lengths in the longitudinal and lateral directions measure approximately 120 by 14 miles, respectively.

The bay can be characterized as having three distinct basins. In the northern basin, extending northward from the Door Peninsula, Green Bay connects with Lake Michigan through four passages. Average water depth in this basin is approximately 65 ft, with depths exceeding 100 ft in the passages. The central basin, extending north from Sturgeon Bay to the Door Peninsula, has an average water depth of approximately 100 ft. With an average depth of approximately 30 ft, the southern basin is considerably shallower than the other basins. The city of Green Bay is located at the southern end of the bay. The Fox River, the major tributary emptying into the bay, passes through this city. Several major industrial and municipal users discharge treated wastewater into the Fox River, and as a result, this river can be classified as polluted (Miller and Saylor 1985).

Transport of pollutants through the river/bay system, together with its wasteload assimilative capacity, is significantly influenced by the interaction of multiple hydrodynamic processes. These include long-term upper Great Lake water level fluctuations, seiche and tidal action, local wind conditions, and Fox River discharges. Local bathymetry, together with the navigation channel, also has a significant influence on current patterns. Having a total length of 14 miles, the channel extends approximately 8 miles into the bay. Relative to the low water datum, channel depths in the vicinity of the Fox River mouth are approximately 22 ft. In contrast, areas adjacent to the channel are much shallower, with depths averaging 5 ft. Furthermore, several areas become exposed during periods of lower lake levels and during extreme seiche events. Thus, the shallow bathymetry, coupled with the multiple forcing mechanisms affecting the lower bay, result in a highly complex system.

The temporal phasing and duration over which these processes interact also influence the degree of DO depletion and the region in which depletion occurs. For example, during storm events, seiche action and/or high river discharges can provide sufficient momentum for transporting the loadings into open water. Because of the oscillatory nature of a seiche, loading transport will alternately be directed upstream, toward DePere Dam, and downstream toward open water.

During calm periods, river discharges may lack sufficient momentum to flush the loadings into open bay areas. Under these conditions, the loading will reside within the river and in the immediate vicinity of the mouth. Should a prolonged calm period occur, the quantity of

loadings may exceed the assimilative capacity of the river, increasing the possibility of anoxic conditions.

Hydrodynamic Model

Water surface levels and velocity fields within the lower bay are simulated with the Curvilinear Hydrodynamic Three-Dimensional (CH3D) model (Johnson et al. 1989). In this application, CH3D was applied in a two-dimensional, vertically averaged (i.e., external) mode. This model possesses the ability to define a basin in boundary-fitted coordinates, allowing coordinate lines to conform with an irregular shoreline and/or channel. Forcing mechanisms incorporated into the CH3D model include water surface level fluctuations, river inflows, surface and bottom shear stresses, and coriolis effect.

The numerical grid is shown as Figure 2. The grid has an overall dimension of 109 by 87 cells. Within the lower bay, omitting the Fox and East Rivers, grid dimensions are 57 by 87 cells. At the northern grid boundary, cell widths in the north-south direction are approximately 1,100 ft, whereas in the east-west direction cell widths range from 100 ft in the vicinity of the channel to 2,600 ft at the grid's western edge. The grid's finest resolution is placed at the Fox River mouth. Cells in this area measure approximately 75 by 125 ft in the east-west and north-south directions, respectively.

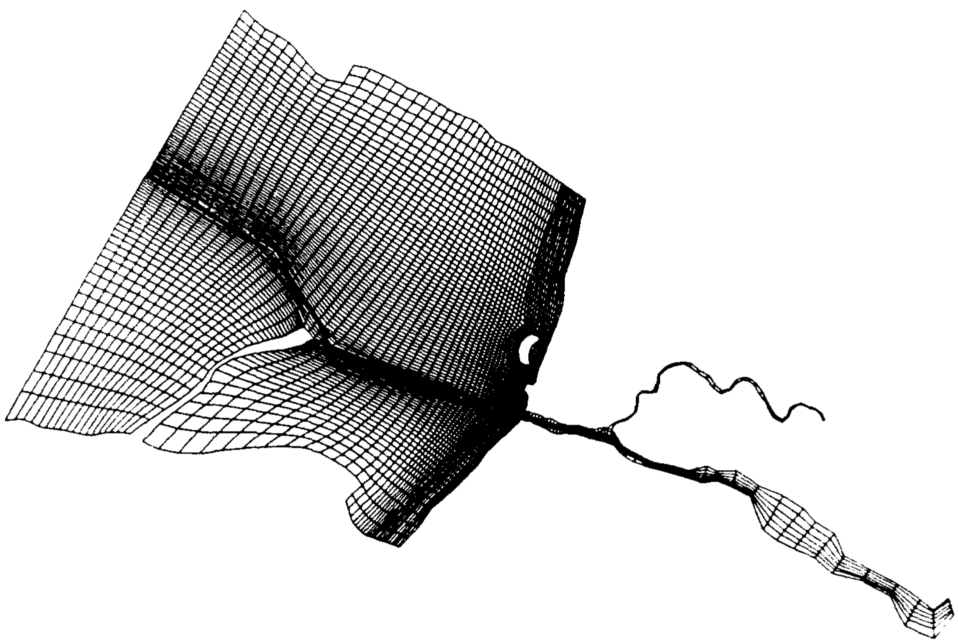

Figure 2. Numerical grid of lower Green Bay

The open-water boundary is driven with a time-series of water surface levels recorded at the Angle Light gaging station. These data were supplied to the model at 15-min intervals. Wind speed and direction time-histories specified in the model were recorded at the Green Bay Municipal Airport, located about 6 miles south-southwest of the bay. Wind data were supplied to the model at 1-hr intervals. Furthermore, wind speeds were increased by 31 percent to account for overland surface friction effects (Patterson 1984).

River discharges were specified at the upstream boundaries of the Fox and East Rivers. Because the U.S. Geological Survey (USGS) does not maintain velocity gages within the study area, it was necessary to substitute flows measured at other locations for these rivers. Flow rates measured at Rapid Croche Dam, located approximately 9 miles upstream of DePere Dam, were substituted for those at this boundary. No adjustments were made to the flow rates measured at the Rapid Croche Dam gage to account for the additional drainage area below the dam. For the East River, flow rates measured by the USGS on the Kewaunee River, located to the east of lower Green Bay, were used as model input. Flow rates were adjusted to account for the differing sizes in drainage areas between these two river basins. Discharge data provided by the USGS consist of daily averaged flow rates.

Calibration of hydrodynamic model

The time span selected for calibrating the hydrodynamic model began on 16 June 1984 at 0000 Central Standard Time (CST) and concluded 14 days later at 2400 CST on 30 June. During this period, the lower bay experienced two relatively high wind events, with one event inducing a maximum range of water level fluctuations of 1.6 ft, or 2.5 times greater than the mean water surface fluctuation range for this area (Patterson 1984).

Data available for comparison with model results include time-series of water surface elevations recorded at the Pulliam Power Plant gaging station, and current speeds and directions from five current meters whose locations are shown in Figure 1. Water levels at the Pulliam Power Plant were recorded at 1-hr intervals. Current data were recorded at 10-min intervals using Endeco 174 shallow-water current meters.

The model accurately reproduced, in both phase and amplitude, the water surface elevation time-histories recorded at the Pulliam gaging station throughout the calibration period. As shown in Figure 3, the model accurately replicated the 12- and 9-hr modes of oscillations. Typically, model-generated water surface elevations differed by less than 0.05 ft when compared with the measured water levels.

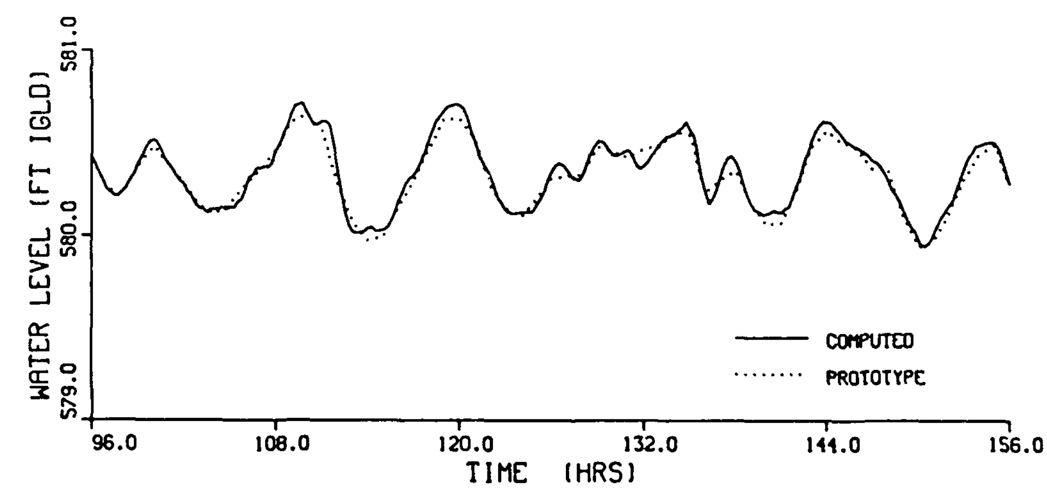

Figure 3. Comparison of water levels at Pulliam gaging station

In general, at the Northeast, North Central, and Southeast current meter locations, model results were in good agreement with the measured current speeds and directions. This

includes both the phases and amplitudes of the lower and higher frequency oscillations recorded by these meters.

Validation and assessment of hydrodynamic model

A 21-day period, extending from 1 July 1984 at 0000 CST to 21 July 1984 at 2400 CST, was selected for validating the model. Two large seiche events were recorded during this period. The most significant event resulted in a water level displacement of 1.6 ft.

Comparisons between the computed and measured water surface levels at the Pulliam gage exhibited the same degree of accuracy as those obtained in the calibration procedure. The model accurately replicated the water surface oscillations, in both phase and amplitude, for the 12- and 9-hr modes of oscillations. In general, predicted water levels were within 0.1 ft of the measured water levels, with extended periods when the differences were less than 0.05 ft. The model accurately matched current speeds and directions together with the amplitudes and phases of the lower and higher frequency oscillations recorded at the North Central and Central Channel current meter locations. Successful calibration and validation demonstrate the model's ability to accurately replicate both measured water surface levels and current speeds and directions. It was therefore felt that the hydrodynamic model could be applied to predict accurate flow fields in lower Green Bay.

Water Quality Model

The three-dimensional integrated compartment model CE-QUAL-ICM was selected for simulating the water quality of the lower bay. The governing equations defining the biological processes can be found in Dortch (1990). Because of the shallowness of the study area, this model was applied in a two-dimensional, depth-averaged mode. A 1:1 overlay of the hydrodynamic grid was used in the water quality model, with the CH3D model supplying the flow field for transporting water quality constituents.

CE-QUAL-ICM contains two sources and five sinks of DO. Sources include reaeration and algal photosynthesis, whereas sinks include sediment oxygen demand, labile and refractory chemical biological oxygen demand, algal respiration, and nitrification. As shown in Figure 4, the model is composed of 11 state variables, one of which is a conservative tracer.

The most severe conditions from a water quality standpoint occur during the summer when water temperatures are relatively high, resulting in increased biological activity and lower DO saturation levels. The water quality model was therefore calibrated to summertime conditions. The most comprehensive data set obtainable was that of the Wisconsin Department of Natural Resources for the summer of 1983. This data set contains both grab sample and continuous monitoring station data collected in the lower bay and along the Fox River. Among the grab sample data were temperature, DO, chlorophyll-a, nitrogen (TKN, NH_3, NO_3), and phosphorus (TP, soluble P). The continuous monitoring data consisted of DO and temperature measurements.

These data were obtained for the entire calibration effort. However, the time span of 13 July through 18 August 1983 was chosen for calibrating the water quality model. The CH3D model provided the hydrodynamic information required by the water quality model.

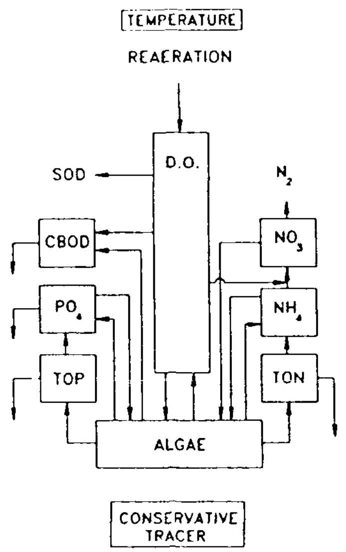

Figure 4. Schematic of lower Green Bay water quality model

Flows and elevations required by the water quality model were updated hourly from CH3D output. Initial conditions were generated from steady-state simulations using constant loadings. Headwater boundary conditions on the Fox River were updated daily, except for DO concentrations, which were updated hourly. Boundary conditions at the northern boundary and the East River were held constant throughout the calibration period. Loadings from the point source dischargers along the lower Fox River were updated daily.

Computed algae concentrations are shown in Figure 5. These concentrations were averaged over the calibration period, and the maximum, mean, and minimum levels computed are plotted against the average and range of the observed data. Both average and extreme computed values compare favorably with the observed data within the lower Fox River but are low near Long Tail Point. It should be remembered that the observed data used for model comparison consist of no more than five measurements at any location. Results show that

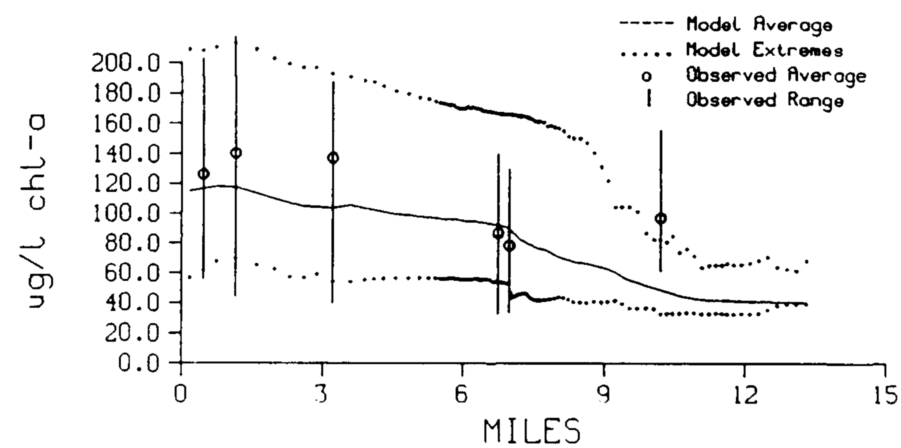

Figure 5. Comparison of computed and measured algae concentrations

algal growth in the model is limited by a combination of low available phosphorus levels and low light intensities due to algal self-shading.

Scenarios

A series of test simulations will be performed to assess whether the proposed expansion of the CDF will adversely impact water quality conditions in the lower bay. This series is composed of two sets of simulations. The first set reflects the existing Kidney Island configuration, whereas the second reflects the proposed expansion. The assessment will be made by comparing DO distributions within the lower bay. Long-term lake levels, riverflow rates, and seiche action/wind events will be treated as independent variables. Seiche action and wind conditions will be treated as one independent variable because a seiche is induced by wind and cannot be readily uncoupled. Other boundary forcing conditions, such as meteorology (e.g., winds, solar radiation), wasteloads, and water quality boundary conditions, will be treated as dependent variables.

Each of the three independent variables can be represented by three levels--high, medium, and low. Thus, a total of 27 simulations, representing both average and higher return period conditions, can be developed if all combinations of the independent variables are considered. This number can, however, be reduced by judiciously selecting those combinations of variables that would cover the range of expected impacts. For example, some events can be eliminated because the combination of independent variables defining those events is implausible. Thus, the number of selected scenarios was reduced to 10, and each scenario will be run with existing geometry and the proposed island expansion (i.e., 20 simulations).

Three time-series of seiche/wind action events will be chosen to represent the three levels of intensity; two will depict historical seiche events, whereas the third will depict a (historical) prolonged calm period. Of the two seiche events, the first will represent an extreme or higher return period storm condition, and the second will represent a seiche commonly experienced in lower Green Bay.

The three selected long-term lake levels are 580.50 ft International Great Lake Datum (IGLD), 578.81 ft IGLD, and 576.93 ft IGLD for the maximum, average, and minimum lake levels, respectively. The average lake level represents the averaged summer mean water level for the 26-year period from 1955 through 1980. Elevations for the high and low lake levels are equal to the average summer mean water level ± 1.5 times the standard deviation in lake levels over this 26-year period.

Time-series of riverflow rates will be selected from the 71-year record of daily averaged flows compiled by the USGS at the Rapide Croche Dam gage. The low-riverflow condition will consist of a time-series of riverflows that contains a 7-day period that approximates the 7-day average low-flow, 10-year return period event (i.e., 7Q10). The "average storm" flow condition will reflect a seasonal high-flow event having a 2-year return period. The maximum riverflow condition will consist of the measured time-series containing the maximum flow rate of record.

Summary

Lower Green Bay is being studied to determine whether a proposed expansion of the CDF will impact water quality in this region. A hydrodynamic model for the lower bay has

been developed, calibrated, and validated. The water quality model has also been calibrated. Presently, simulations are being performed with pre- and post-expansion CDF configurations under a variety of scenarios. Following completion of these scenarios, an assessment will be made whether the proposed expansion adversely affects DO concentrations in the study area.

Acknowledgments

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Application of the CE-QUAL-ICM Eutrophication Model to Chesapeake Bay

by
Carl F. Cerco[1] and Tom Cole[1]

Introduction

We are nearing completion of a 3-year study to develop and apply a model of eutrophication processes in Chesapeake Bay. Capacities of the model include analysis of the origins and extent of eutrophication in the Bay, prediction of future water quality conditions as a result of management action or inaction, and reconstruction of processes that led the Bay to its present condition. Complete documentation of the model and its application to the Bay is under way (Cerco and Cole, in preparation). This paper outlines the model structure and provides examples of model applications including system mass balances, scenario analyses, and historical reconstructions.

The Eutrophication Model

CE-QUAL-ICM is one component of a package of models applied to Chesapeake Bay. Other components of the package are the CH3D hydrodynamic model (Johnson et al. 1991) and a predictive sediment oxygen demand and benthic nutrient flux model (HydroQual, Inc. 1991). CE-QUAL-ICM is an integrated compartment box model. Water quality in each box is computed by numerical integration of a system of mass-conservation equations that account for transport across compartment boundaries, external loading, and internal sources and sinks. The box structure was selected to allow maximum flexibility for adaptation of the model to alternate hydrodynamic models.

The eutrophication process is modeled with 22 variables that are categorized into several groups or cycles. The physical group consists of salinity, temperature, and inorganic suspended solids. Salinity is included primarily to verify the linkage to CH3D; when the models are properly linked, salinity predicted by the water quality model replicates predictions by CH3D. Temperature is included for its fundamental impact on the rate of biochemical processes. In the present model configuration, inorganic solids are used as a surrogate for iron and manganese that sorb PO4 and dissolved silica (DSil) in the water column.

The model carbon cycle (Figure 1) originates with carbon fixation by three algal groups. Algal carbon is respired away or converted to dissolved organic carbon (DOC) or particulate organic carbon through algal mortality and predation. Two particulate carbon groups, labile (LPOC) and refractory (RPOC), are included to reflect the differential rates at which organic matter decomposes. A fraction of the particulate carbon is hydrolyzed to DOC. The balance of the particulates, plus a portion of the algae, settles to the bottom where the material is incorporated into the sediments. DOC produced by algae or by hydrolysis of particulates is removed from the system through the process of heterotrophic respiration.

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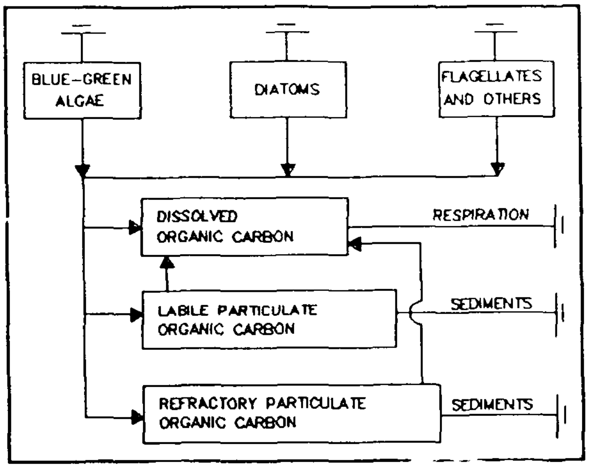

Figure 1. The carbon cycle

In the model nitrogen cycle (Figure 2), nitrate (NO3) and ammonium (NH4) are incorporated by the three algal groups. The actions of mortality and predation release algal nitrogen back to the water column as dissolved organic nitrogen (DON), labile and refractory particulate organic nitrogen (LPON, RPON), and as NH4. Hydrolysis converts a portion of the particulate nitrogen to DON. The remainder settles into the benthic sediments along with the nitrogen incorporated in the biomass of settled algae. DON is mineralized to NH4 and rendered available again to the algae either as NH4 or as NO3 produced by nitrification.

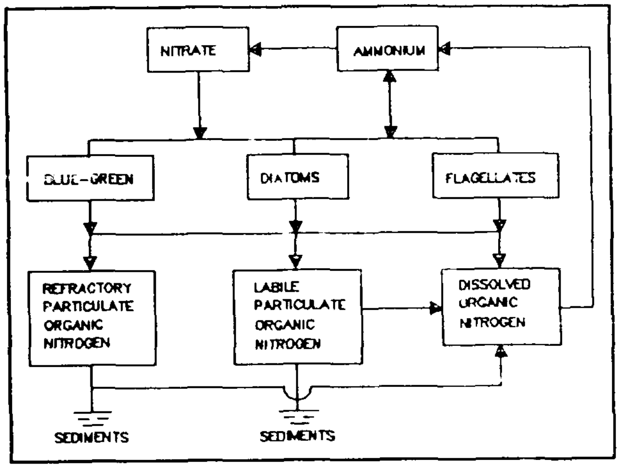

Figure 2. The nitrogen cycle

The model phosphorus cycle (Figure 3) is similar to the nitrogen cycle. Phosphate is incorporated by three algal groups and released as dissolved organic phosphorus (DOP), labile and refractory particulate organic phosphorus (LPOP, RPOP), and as PO4. A portion of the particulate phosphorus is hydrolyzed to DOP; the balance is lost to the sediments along with phosphorus incorporated in settled algae. DOP is mineralized to PO4 and made available again to the algae. The phosphorus cycle differs from nitrogen, however, in that PO4 undergoes sorption-desorption with iron and manganese particles. This process is included based on observations that sorption and subsequent settling is a significant pathway for removal of phosphorus from the water column.

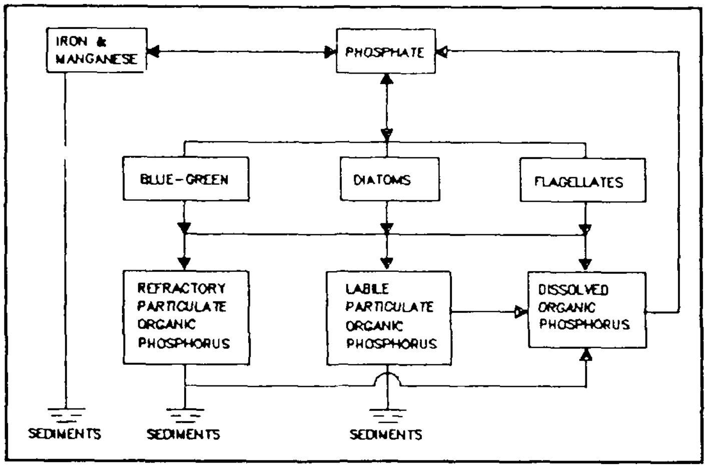

Figure 3. The phosphorus cycle

The model silica cycle (Figure 4) is a simple one. Dissolved silica is incorporated by diatoms and released to the water column through the actions of predation and mortality as DSil or particulate biogenic silica (PBS). The PBS undergoes dissolution in the water column or else settles into the benthic sediments along with the silica bound in the biomass of settling diatoms. As with PO_4, DSil undergoes sorption interactions with iron and manganese particles.

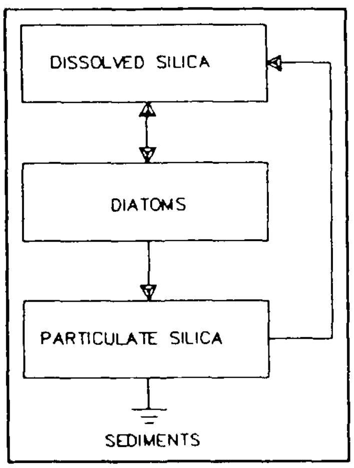

Figure 4. The silica cycle

The remaining cycle in the model is the dissolved oxygen (DO) cycle. Sources of DO include algal photosynthesis and atmospheric reaeration DO is lost through algal and heterotrophic respiration and through exertion of chemical oxygen demand (COD) released from the sediments.

Application to Chesapeake Bay

The Chesapeake Bay system (Figure 5) consists of the main stem Bay, five major western-shore tributaries, and a host of lesser tributaries and embayments. The main stem is roughly 300 km long, 8 to 48 km wide, and 8 m in average depth. A deep trench with

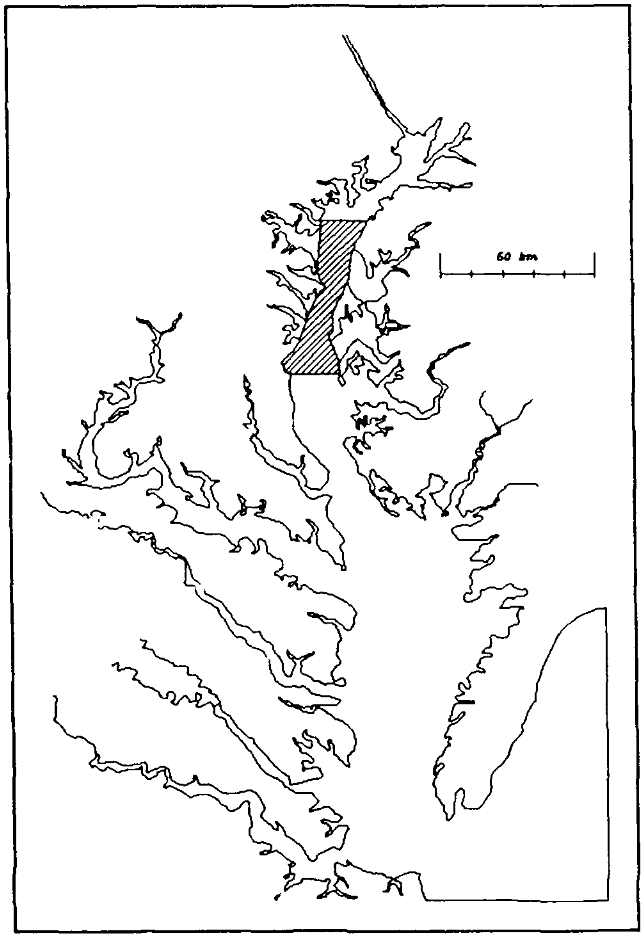

Figure 5. Chesapeake Bay

depths to 50 m runs up the center of the main stem. The primary source of fresh water to the system is the Susquehanna River (≈62 percent of total gaged freshwater flow), which empties into the northernmost extent of the Bay. The Bay is a classic example of a partially mixed estuary in which long-term average circulation is upstream along the bottom and downstream near the surface, although this pattern is frequently altered by local and distant meteorological events. Major urban centers along the Bay and tributaries include Norfolk and Richmond, VA, Washington, DC, and Baltimore, MD.

The model was calibrated through application to the period 1984-1986. The model was run continuously through the 3-year period, and employed CH3D hydrodynamics generated for each year and initial conditions generated by a long-term run of the water quality model. The primary database for model calibration and performance evaluation was provided by the EPA's Chesapeake Bay Monitoring Program. The program conducted 20 Bay-wide surveys per year that sampled roughly 90 stations in the main stem Bay and five major tributaries.

Mass Balance and Fluxes

The mass balance is a common analytical tool employed in examining estuarine biogeochemical processes. Mass balances are performed both as a check on model computational accuracy and to gain insight into Bay behavior. A balance on water column carbon

(Figure 6), for example, indicates that net algal production is the largest source to the system. The primary sink of carbon is the sediments in which carbon is oxidized creating a DO deficit in the water column. An attempt to control carbonaceous oxygen demand by control of external carbon discharges to the Bay would reduce only a fraction of the total carbon entering the system. This insight reinforces the strategy of controlling oxygen demand in the system by reducing the nutrients that support algal production.

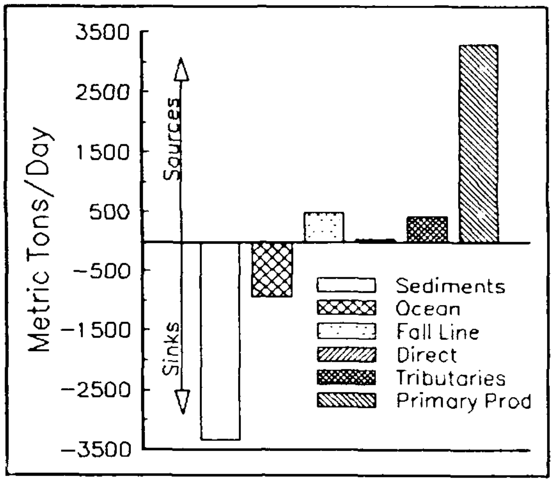

Figure 6. Water-column carbon balance

Flux analyses of nitrogen (Figure 7) and phosphorus (Figure 8) in the main stem Bay indicate the Susquehanna fall line is the major source of these nutrients. For nitrogen, the contribution from tributaries is the next largest source followed by direct point- and nonpoint-source loads. Nitrogen is lost from the Bay to the ocean. By contrast, phosphorus is imported from the ocean in quantities comparable to tributary and direct inputs. Differences in the magnitude and controllability of nitrogen and phosphorus sources provide insights to the feasibility of various nutrient control str tegies for the Bay.

Scenario Analyses

Scenario analysis is perhaps the most familiar form of model application. In scenario analysis, the model is used to predict the effect on a water body of alternate management actions in combination with various environmental influences. Two suites of 10 sensitivity scenarios have been performed for the Bay. Another suite of 25 is under way. A total of 100 to 200 scenarios is anticipated.

A primary goal of management activity in the Bay is elimination of anoxic bottom water. Scenarios run so far have examined the response of the Bay for conditions ranging from nutrient load increases to feasible control actions to restoration of pristine conditions. Preliminary results indicate, in an average year, that anoxia did not exist under pristine conditions (Figure 9). Management alternative A achieves the goal of elimination of anoxia while alternative B fails in this regard.

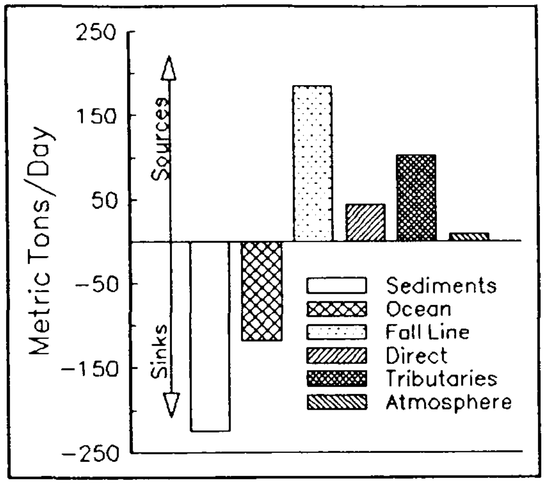

Figure 7. Water-column nitrogen balance

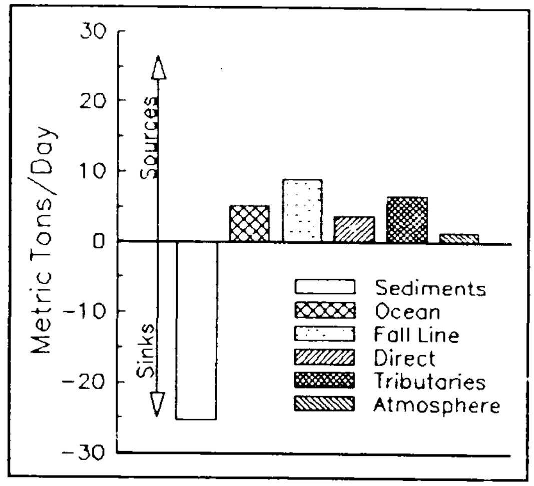

Figure 8. Water-column phosphorus balance

Historical Reconstructions

Water quality in Chesapeake Bay varies from year to year. Superimposed upon the inter-annual variability, a long-term trend toward degradation has been suggested by some researchers (Heinle et al. 1980, Officer et al. 1984). The issue of long-term trends is controversial, however. First, trends are difficult to detect in the historic data record because of gaps and variations in sampling frequency and location. Second, trends must be isolated from periodic variations in loading and hydrodynamics that occur on time scales ranging from days to decades. The model is an ideal tool for examining the issue of long-term trends since it allows the isolation of deterministic events from random variability. Model sensitivity analysis allows manipulation of factors that influence water quality in order to determine the

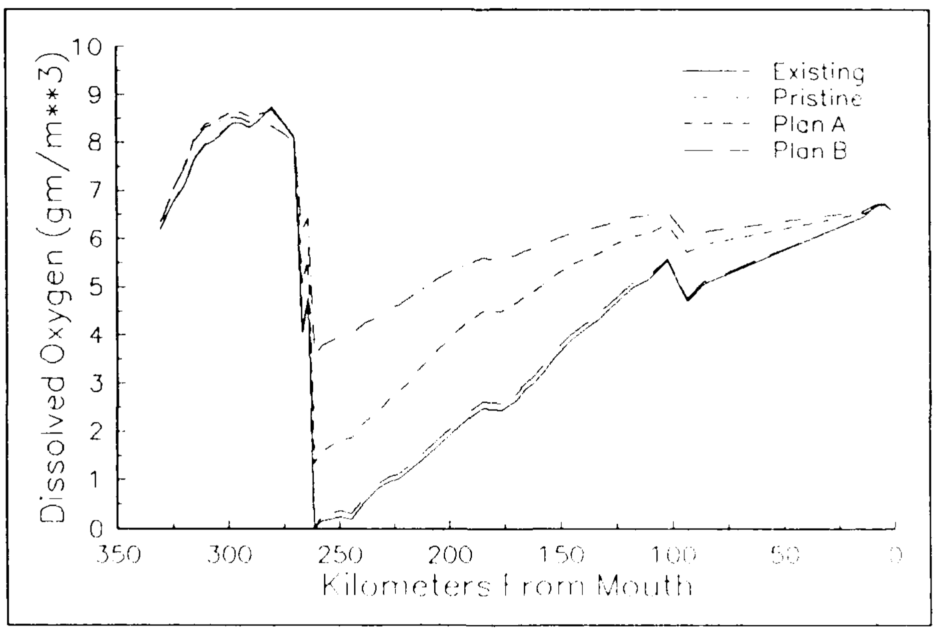

Figure 9. Bottom dissolved oxygen in average year

primary factors that contribute to trends. We have completed several 30-year runs aimed at reproducing trends in Chesapeake Bay DO from 1959 to 1988. We believe these are the most extensive estuarine simulations yet conducted.

Initial application of the model to the examination of long-term trends in DO has been promising (Figure 10). The water quality model exhibits no drift or aberrant behavior. Runs initiated in 1959 converge upon observations collected at the commencement of the Bay monitoring program in early 1984. Dissolved oxygen predictions in the upper Bay indicate that anoxia is more prevalent in wet years than in dry to average years. This tendency coincides with an interpretation of long-term trends proposed by Seliger and Boggs (1988). Demonstration of model-data agreement with the historical record is inconclusive, however. We are presently focusing our efforts on alternate examinations and treatments of the data. Sensitivity analyses of the model predictions to loads and hydrodynamics are planned, as is examination of model output for long-term trends.

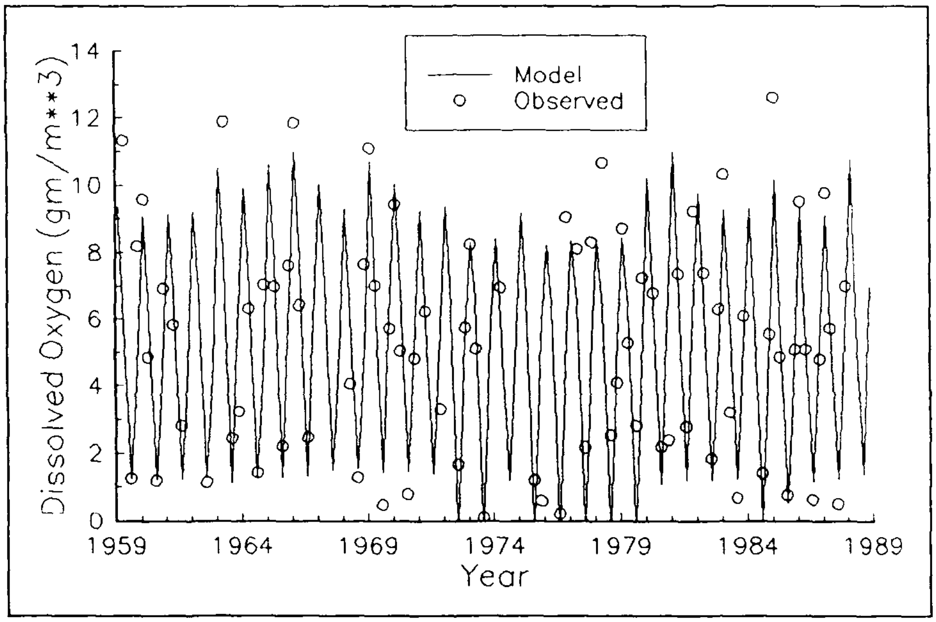

Figure 10. Thirty-year time-series of bottom dissolved oxygen (data and model from shaded area, Figure 5)

Conclusion

We have demonstrated some of the capabilities of CE-QUAL-ICM through application to Chesapeake Bay. Employment of CE-QUAL-ICM is not limited to estuaries, however. Formulation of the model allows for one-, two-, or three-dimensional application to lakes, rivers, and coastal waters. The model is presently being applied to the Green Bay embayment of Lake Superior, to the Indian River-Rehoboth Bay Estuary system, and to the New York Bight.

Acknowledgments

The Chesapeake Bay model package is being developed as part of a program sponsored by the U.S. Army Engineer District, Baltimore, and a state-Federal partnership administered

by the Chesapeake Bay Liaison Office, Region III, U.S. Environmental Protection Agency. Permission was granted by the Chief of Engineers to publish this information.

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Dissolved PCB in the Saginaw Confined Disposal Facility During Dredged Material Disposal

by
Tommy E. Myers[1] and Pam Bedore[2]

Abstract

Concentrations of dissolved polychlorinated biphenyls (PCBs) in pond water samples from the Saginaw confined disposal facility (CDF), Bay City, MI, were measured during disposal of dredged material from the Saginaw River near Saginaw, MI. These concentrations were compared with their concentrations in the dredged material influent to obtain estimates of containment efficiency. Effluent monitoring was not practical because the perimeter dikes at the Saginaw CDF are permeable, and discharge through permeable dikes is a diffuse source that is quickly diluted to background concentrations. Consequently, pond water samples provided the best approximation to the effluent.

Two procedures for predicting dissolved PCB concentrations in CDF pond water were evaluated--the modified elutriate test and equilibrium partitioning calculations using site-specific distribution coefficients. Predictions from the modified elutriate test and equilibrium partitioning calculations were similar and generally within a factor of 2 to 4 of the observed concentrations.

(The tests described and the resulting data herein, unless otherwise noted, were obtained from research conducted by the U.S. Army Engineer Waterways Experiment Station for the U.S. Army Engineer District, Detroit. Permission to publish this information was granted by the Chief of Engineers.

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[2] U.S. Army Engineer District, Detroit; Detroit, MI.

Bioaccumulation Evaluation of Sediments Using the Guidance Provided in "Evaluation of Dredged Material Proposed for Ocean Disposal" ("The Green Book")

by
Francis J. Reilly, Jr.,[1] Victor A. McFarland,[2]
Joan U. Clarke,[2] A. Susan Jarvis,[2]
Charles H. Lutz,[2] and J. Brian Mulhearn[1]

Abstract

Several methods for assessing the potential for bioaccumulation are included in the Green Book. Tier II evaluation involves calculating the potential for bioaccumulation from sediment data. Tier III involves performing actual exposures to demonstrate bioaccumulation, and Tier IV involves the performance of special tests to predict the steady-state concentration of contaminants in aquatic organisms. It is not necessary to perform all the tiers for any given sediment or project.

The Environmental Laboratory of the U.S. Army Engineer Waterways Experiment Station recently evaluated test methods for bioaccumulation of contaminants in marine organisms. Several sediments (the San Francisco Bay region/Oakland, Norfolk/Hampton Roads, and New York Bight) were used to expose a variety of appropriate organisms to both bedded and suspended sediments according to the procedures detailed in the Green Book. The organisms selected were *Macoma nasuta* and *M. secta*, two species of deposit-feeding clams, chosen for their inability to metabolize polycyclic aromatic hydrocarbons; a large marine deposit-feeding worm, *Nereis virens*; the blue-mussel, *Mytilis edulis*, a filter-feeder that is widely employed as a contaminant monitor; and the sand dab, *Citharichthys stigmaeus*, a flatfish that is in close contact with the sediment. Test organisms were exposed for either 10 or 28 days in a flow-through apparatus, and contaminant residues were measured. Results showed that organisms exposed using the guidelines in the Green Book do bioaccumulate contaminants from sediments, and that it may be possible to predict a priori the levels of neutral organic compounds that will be taken up by organisms exposed to contaminated sediments.

This paper examines the reasoning behind bioaccumulation testing, the actual practice of testing, and considers the usefulness of data obtained from bioaccumulation testing.

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Application of Simulated Acid Rain As an Alternative TCLP Extraction Medium

by
Frank Snitz[1]

Abstract

The Michigan Department of Natural Resources (MDNR) (Act 641) prescribes the application of Toxicity Characteristics Leaching Procedure (TCLP) protocol to characterize dredged material suitability for upland disposal. The MDNR has upper pass/fail concentrations in the TCLP media and lower values. The designation "greater than upper limits" indicates hazardous and toxic waste, while "less than lower limits" means "inert." Intermediate values indicate conventionally contaminated material, suitable for disposal in a normal licensed "Class II" landfill.

St. Joseph River dredged material was subjected to TCLP protocol and found to exceed the lower limit. This meant expensive engineering of the proposed Thar Road upland disposal site, a geotextile liner, etc.--at an additional cost of more than $10 million. Because the TCLP extraction medium fluid 2, at a pH of 3, is considerably more aggressive than site conditions could ever conceivably be, we proposed a more realistic worst-case extraction medium--one that would simulate acid rain. We proposed using distilled water acidified with H_2SO_4 to a pH of 4.

Application of this Simulated Acid Rain extraction fluid in the TCLP protocol produced concentrations of heavy metals that were below the MDNR "Inert" category limits, thus suggesting, at this preliminary stage, that the Thar Road disposal site may be made suitable with only modest levels of site excavation.

[1] U.S. Army Engineer District, Detroit; Detroit, MI.

An Evaluation of Sediment-Rela*ted Factors Influencing Wintertime Dissolved Oxygen Levels in the Big Eau Pleine Reservoir, Wisconsin

by
Douglas Gunnison,[1] William F. James[1]
Harry Eakin,[1] and John W. Barko[1]

Introduction

In 1937 the Wisconsin Valley Improvement Company constructed the 2,760-ha Big Eau Pleine Reservoir by impounding the Big Eau Pleine River. The reservoir was created to assist in providing uniform flows in the Wisconsin River, and has had a subsequent history of major winter fish kills and summer algal blooms (Gunnison and Barko 1990). Extensive studies of the reservoir were conducted between 1974 and 1979 to evaluate sources of water quality problems, and to suggest possible management options for improving general environmental conditions (Shaw and Powers, undated). Based on the results of these earlier studies and additional data, Gunnison and Barko (1988, 1990) proposed the following hypothetical scenario of events leading to reductions in dissolved oxygen (DO) during the winter in the Big Eau Pleine Reservoir. Development of anoxia in the reservoir results from sediment-associated oxygen sag formation in the headwaters during winter drawdown, coupled with low inflows. Continued withdrawal from the reservoir enhances entrainment of anoxic water and its downstream movement. Anoxic waters originating in the headwaters often merge with hypolimnetic pockets of anoxia near the dam, involving extensive areas of the reservoir in low DO conditions by late winter.

Objective

The primary objective of the most recent studies (1989-1991) conducted by the Waterways Experiment Station on the Big Eau Pleine Reservoir was to evaluate, to the extent possible, the hypothetical scenario leading to decreases in DO in the Reservoir during the winter. These studies were conducted during spring periods of high runoff and sediment transport, and in the winter beneath the ice during periods of reservoir drawdown and sediment refocusing. These studies were designed to address seasonal changes in sediment physical and chemical composition and effects on water quality of exchanges from surficial sediment.

Materials and Methods

Sediment samples were collected in May 1989 and January and May 1990 for purposes of determining physical and chemical composition of the sediment and interstitial water. Stations were located at approximately 1-mile intervals, or less, along the longitudinal axis of the reservoir between river miles (RM) 0.3 and 18.3 (Figure 1). With three exceptions, stations along this axis were positioned laterally at the point of maximum depth (i.e., within the

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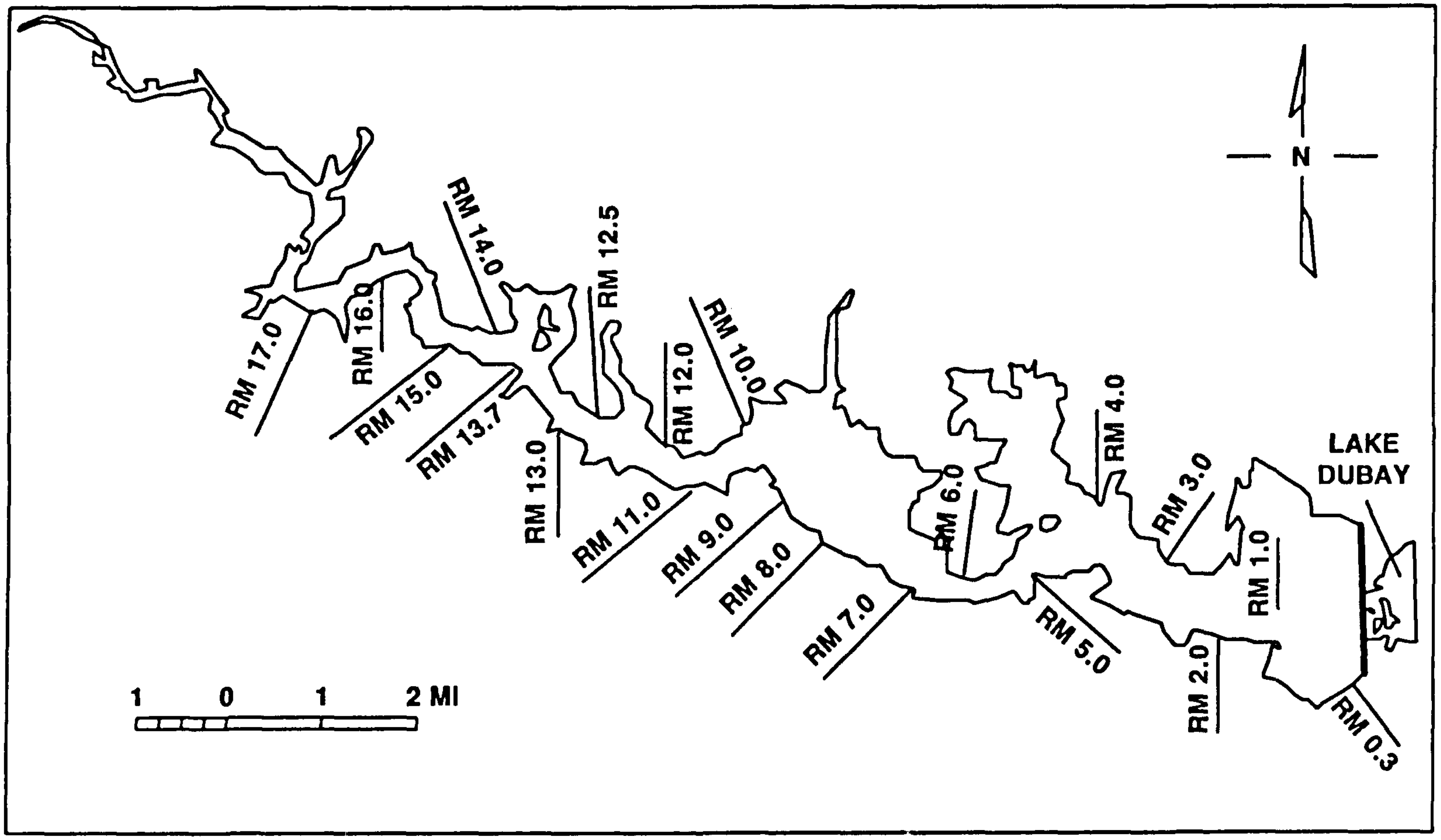

Figure 1. Big Eau Pleine Reservoir

thalweg). Sampling stations outside the thalweg at RM 0.3, 4, 8, 11, 12, 13, and 16 were located along transects that ran perpendicular to the shoreline. Sampling stations were located during successive sampling periods to within 5 m by theodolite measurement of angles to the sampling location, and then electronic measurement of distances to each location. In May of both years, thalweg and nonthalweg stations were sampled. However, samples could not be collected between RM 1 and 7 in May 1989, as the result of difficulties in obtaining core samples in flocculent sediments. Winter drawdown restricted sampling to stations located only in the thalweg in January 1990.

Sediment cores were collected using a 2-in. corer. Cores were sealed, stored vertically at 4 °C in the dark, and transported intact to the laboratory. Cores were sectioned at 5-cm intervals, and processed for moisture, density, and organic matter determinations. Surficial sediments were collected using a 15- by 15-cm grab sampler. Samples were held under nitrogen. Grab samples were homogenized and analyzed for moisture, density, organic matter, and sediment chemical oxygen demand (COD). Interstitial water was separated from sediment by centrifugation under nitrogen, and then analyzed for conductivity, soluble reactive phosphorus (SRP), ammonium-N, dissolved organic carbon (DOC), COD, and soluble iron and manganese.

Results

Moisture content of the surficial sediment was greatest in deep regions of the reservoir, and decreased with decreasing water depth. Based on Håkanson's (1977) relationships between energy environments and moisture contents, three zones of sedimentation were identified (Figure 2): an erosional zone (moisture content <50 percent), a transport zone (moisture content 50 to 75 percent), and an accumulation zone (moisture content >75 percent). Mean moisture content was generally less than 50 percent in the headwaters of the reservoir (RM 12-17), indicating that erosional forces were acting on this region. A region of sediment transport between RM 7 and 11 was indicated by moisture contents of 50 to 75 percent, and a zone of sediment accumulation (moisture content >75 percent) was observed between RM 0 and 6.

Sediment density was inversely related to moisture content, thus showing trends in relation to depth opposite to those of moisture content. Sediment density increased with increasing depth (Figure 3) and increasing distance from the dam. In the thalweg, sediment densities were high between RM 13 and 17 in May 1989, compared to those levels near the dam. Sediment densities also decreased markedly at these river miles in January 1990, approaching concentrations measured near the dam. In May 1990, peaks in sediment density occurred in the headwaters at RM 14 and 17.

Sediment organic matter content was greatest in deep regions and least in shallow headwater regions of the reservoir (Figure 4). While no sediment organic matter data were obtained at RM 0-7 in May 1989, variations in this parameter between other sampling dates were minor over these river miles in comparison to variations occurring in upper regions of the reservoir. Generally, organic matter concentrations between RM 14 and 17 were greatest in May 1989 and 1990 and lowest in January 1990.

Stations located in the thalweg demonstrated sediment and interstitial water COD concentrations with pronounced peaks of 160 mg/g dry sediment and 290 mg/L, respectively, during

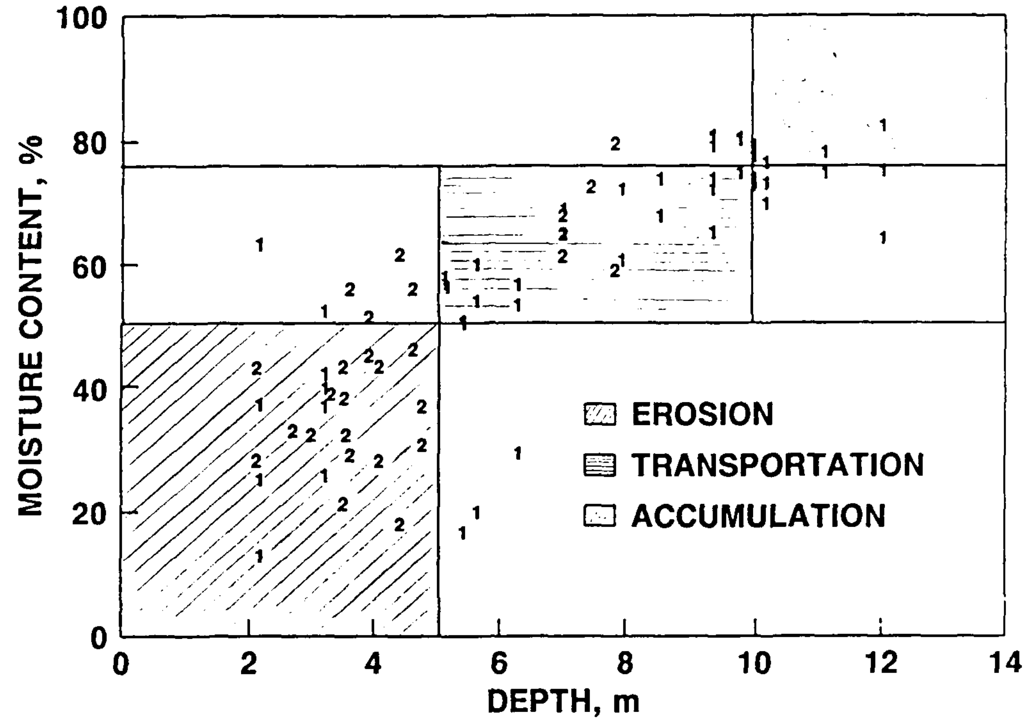

Figure 2. Variations over all dates in moisture content of the surficial sediment with depth for stations located both within and outside the thalweg. Zones of sediment erosion, transport, and accumulation are estimated based on moisture content (see text). (1 = thalweg, 2 = nonthalweg)

May 1989 at RM 10. There was a second interstitial water COD peak of 220 mg/L at RM 11 during the May 1989 sampling period Downstream values of sediment COD were also elevated during May 1989. Particularly high levels of 140, 140, and 138 mg/g dry weight sediment were at RM 1, 5, and 6, respectively, while slightly lower values of 100, 80, 110 mg/g dry weight sediment were at RM 2, 7, and 8, respectively. Interstitial COD also showed relatively high values in the range of 130 to 185 mg/L during May 1989 at downstream locations between RM 3 and 7. Concentrations of sediment and interstitial COD were lowest in the headwater region (i.e., >RM 11), averaging less than 80 mg/L during this period. Sediment interstitial COD values reached their maximum value at RM 10 during May 1989.

During January 1990, concentrations of both sediment COD and interstitial COD decreased to average values of approximately 75 mg/g dry weight and 170 mg/L, respectively in the headwaters, as peaks in concentration diminished near RM 10 from values observed in May 1989. In May 1990, all sediment COD values were considerably less than those observed in January 1990. Decreases in interstitial COD were observed between May 1989 and January 1990, and also between January 1990 and May 1990, at RM 4, 5, 6, 8, 10, and 11. However, increases in interstitial COD were observed between May 1989 and January 1990, followed by decreases to or near the May 1989 levels by May 1990 at RM 0.3, 1, 2, 3, 7, 12, and 16. No explanations for this erratic behavior are suggested by the data, but the general magnitude of the changes observed suggests that some sediment disturbance may have occurred, especially between RM 10 and 13. Interstitial levels of iron, manganese, and

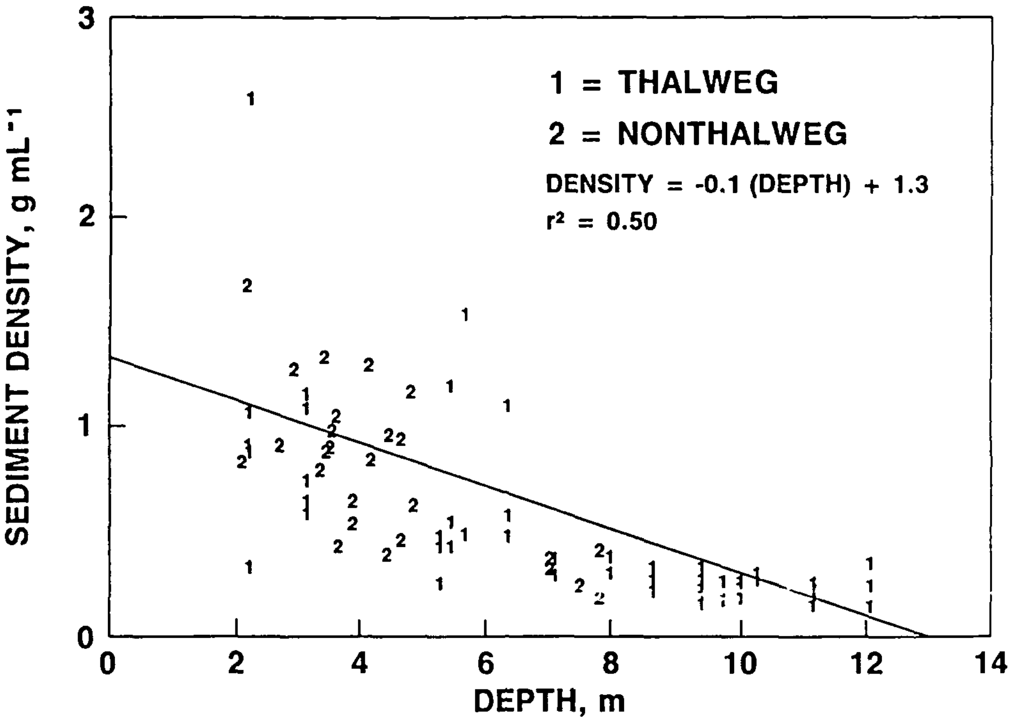

Figure 3. Variations over all dates in density of the surficial sediment with depth for stations located within and outside the thalweg (1 = thalweg, 2 = nonthalweg)

conductivity also showed distinct peaks in the thalweg at RM 10 and lower levels in the headwaters in May 1989 (James et al. 1992).

DOC in the interstitial water exhibited an extensive peak between RM 10 and 13 in May 1989. The peak in interstitial DOC of 87.2 mg/L at RM 10 was more than twice the magnitude of concentrations over the first 8 miles in May 1989. Interstitial DOC decreased substantially in January 1990 at RM 10-13, and concentrations were similar between stations along the longitudinal axis of the reservoir in May 1990. These seasonal variations in interstitial DOC, as well as in iron, manganese, and conductivity, indicate that significant releases of these dissolved materials may have occurred sometime between May and January in the RM 10 area.

Discussion

Sediment composition, resuspension, and transport

Based on physical composition of the sediment, it is apparent that sediment erosion and, therefore, potential resuspension during the winter is pronounced in the Big Eau Pleine Reservoir. This is particularly true in the headwaters, where resuspension is likely followed by displacement and transport downstream to deeper regions near the dam. Based on other work presented by James et al. (1992), net sedimentation rates are relatively low in the headwaters, even though loading to the Big Eau Pleine Reservoir is high (Shaw and Powers, undated;

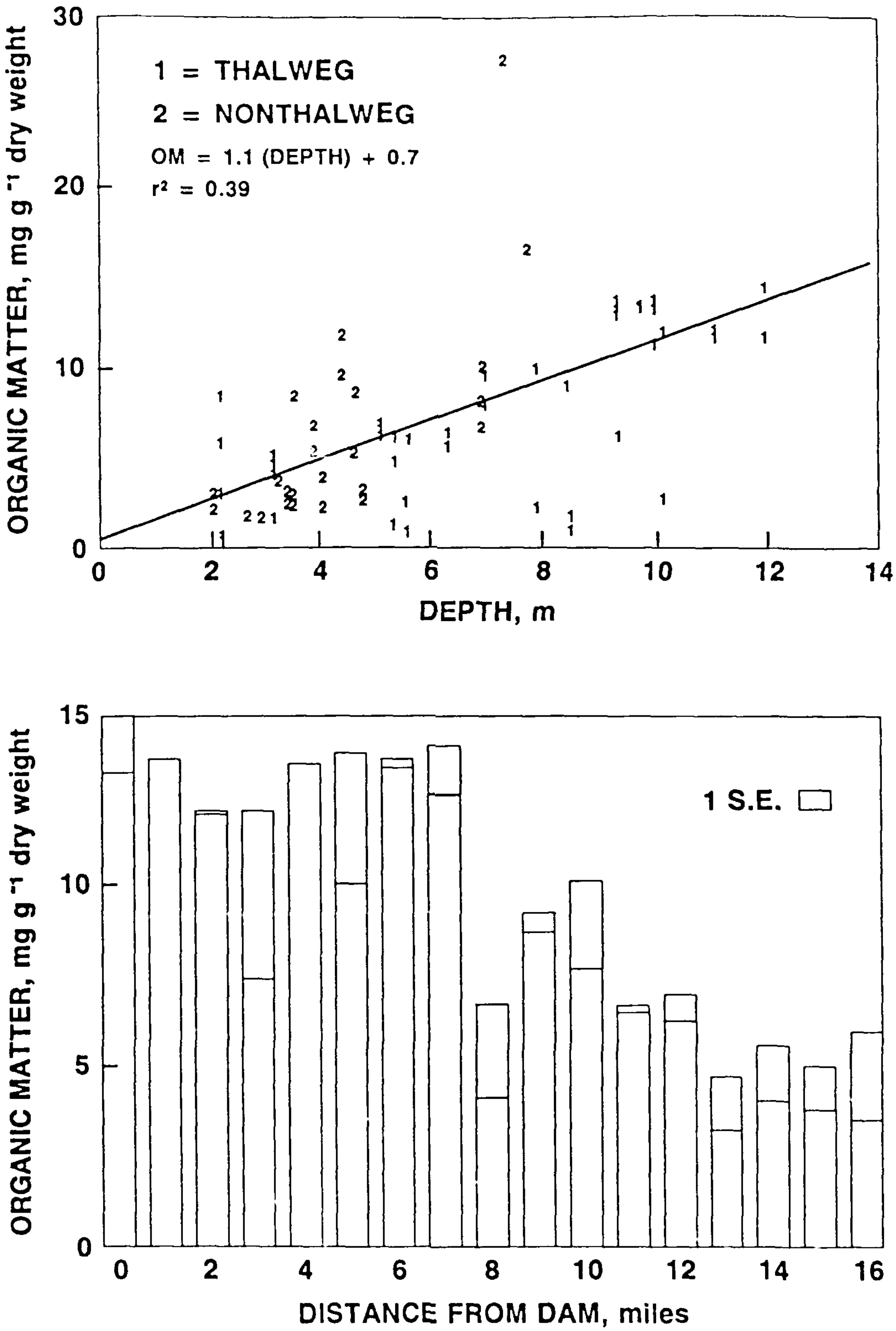

Figure 4. Variations in mean (± 1 S.E.) organic matter of the surficial sediment with distance from the dam for stations located in the thalweg

Gunnison and Barko 1988). Moisture content is also very low (< 50 percent), while sediment density is high in the headwaters, indicating the probable action of erosional forces that resuspend and remove fine sediment from this region (e.g., Håkanson 1977). This sedimentary pattern is unusual for reservoirs (see Thornton et al. 1981), where high sedimentation rates (James et al. 1987) and sediment delta formation (Pharo and Carmack 1979) have been observed in the upper regions near tributary inflows.

Investigations in both lakes and reservoirs have shown that erosional forces (e.g., wave scouring) and basin morphometry result in the focusing of sediment to deep basins (Likens and Davis 1975; Håkanson 1977; Evans and Rigler 1983; Hilton 1985; Hilton, Lishman, and Allen 1986; James and Barko 1990). In the Big Eau Pleine Reservoir, sediment disturbance is evident in the deep thalweg of the headwater region, as well as at shallow depths throughout the reservoir. Our results suggest that sediment loads deposited in the headwaters are being disturbed in the winter, and potentially transported along the longitudinal axis of the reservoir to the deeper basin near the dam. The seasonal variability of sediment moisture content, sediment density, and organic matter in the headwater sediments, compared with the levels of these parameters in sediments located near the dam, also suggests that sediments are being deposited and removed from the headwater region. The large area (60 percent of the reservoir surface) occupied by the erosional zone is within the range of the region of average maximum drawdown for the years 1974-1990 (mean = 1,128 ft MSL), suggesting that winter drawdown and spring refilling may be the principal factors in disturbance of sediment and longitudinal transport toward the dam.

Sediment COD and interstitial water quality

Interstitial water of the surficial sediment in the headwaters exhibited moderately high concentrations of iron and manganese. Both sediment and interstitial water COD were highly correlated with concentrations of iron and manganese in the interstitial water, indicating that these metals may have provided an important source of COD to the water column during reservoir drawdown and sediment disturbance. Gunnison and Barko (1990) proposed that mobilization of these metals and other sources of COD from the sediment into the water column via sediment disturbance and resuspension, diffusion, and/or ice scouring could result in the immediate removal of over one-half the dissolved oxygen, at saturation levels (13.1 mg/L at 4 °C) in the overlying water column.

Throughout the reservoir, sediment COD and interstitial COD, iron, and manganese were generally much less variable along the longitudinal axis during the winter of 1990 than in either spring 1989 or 1990. In particular, peaks in concentrations of these variables at mid-reservoir locations observed in May 1989 had diminished by January 1990. These marked seasonal variations between sampling dates suggest mobilization of sedimentary COD during winter drawdown and transport downstream. Sediment disturbances during winter drawdown appear to be important mechanisms in the mobilization and transport of COD. Increased flow velocities in the headwaters, as a result of decreased channel width during drawdown, may promote sediment disturbance. Diffusion of reduced chemical species out of the sediment is probably enhanced by these same processes. Ice scour and shelving during drawdown may also disturb sediments, resulting in mobilization of COD.

Conclusions

The results of these investigations support the hypothetical scenario of events leading to depletion in dissolved oxygen during the winter in the Big Eau Pleine Reservoir (Gunnison and Barko 1988). Dissolved oxygen depletion and anoxia in the headwaters of the Big Eau Pleine Reservoir are associated with winter drawdown, coupled with low inflows. Dissolved oxygen depletion and anoxia also occur in the hypolimnion near the dam. Entrainment of anoxic water and its downstream movement during winter are promoted by continual withdrawal from the reservoir. Anoxic waters originating from the headwaters often converge with hypolimnetic pockets of anoxia near the dam, forming an extensive region of low dissolved oxygen by late winter.

The composition of the surficial sediment and net sedimentation rates indicate substantial disturbance of sediment in the headwaters and sediment focusing to the deep basin, near the dam of the reservoir. Physical sediment composition varies greatly at different times of the year in the headwaters, suggesting deposition, sediment disturbance, and removal of sediment.

Interstitial water concentrations of soluble iron, manganese, DOC, and COD are moderately high in the surficial sediments of the headwaters and have the potential to exert a substantial COD to the overlying water column, if sediments are resuspended or scoured by ice movement. High levels of phosphorus and suspended solids in the water column during winter drawdown (James et al. 1992) suggest possible sediment disturbance and resuspension. However, the exact mechanisms affecting sediment resuspension during winter drawdown are currently unknown. Interstitial water concentrations of soluble iron, manganese, DOC, and COD, while somewhat lower in the downstream reaches of the project, also have the potential to exert a COD on the overlying water column. Since sediments in the downstream areas do not appear to be resuspended during drawdown, the release of oxygen-demanding materials from these sediments is probably limited to diffusion alone. Nonetheless, significant downstream oxygen depletion can occur, particularly near the dam.

Acknowledgments

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Sediment Oxygen Demand and Its Effects on Water Quality in Corps of Engineers Projects

by
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Introduction

Interactions between bottom sediments and overlying waters are often dominant factors contributing to poor quality of surface waters. Sediment oxygen demand (SOD) is a key contributor to low dissolved oxygen (DO) levels in the water column (Hargrave 1972, Giga and Uchrin 1990). SOD is the rate of oxygen removal from the water column due to the decomposition of settled organic matter and is dependent upon the microbial and chemical oxidation activity in the sediment. Dissolved oxygen concentrations and SOD-regulated processes at the sediment-water interface regulate the movement of reduced chemical species from the sediment surface to the overlying water.

Many nutrients and heavy metals have a high affinity for sediment. Movement of these materials from the sediment to the overlying water is related to SOD processes occurring at the sediment-water interface (Brannon et al. 1983; Gunnison, Chen, and Brannon 1983). Microbial degradation of organic matter in the sediment results in a depletion of DO and release of nutrients and metals, such as nitrogen, phosphorus, ammonium, iron, and manganese, into the water column. The development of anoxic conditions combined with the release of potentially toxic nutrients and metals can lead to severe water quality problems (Gunnison, Chen, and Brannon 1983).

Predicting the effects of U.S. Army Corps of Engineers (CE) water resource projects on water quality has been difficult because of the lack of methods to accurately quantify SOD and related processes. An understanding of SOD processes and their quantification is needed to assess the effects of CE projects on sediment-water interactions and water quality. Currently, no standard method to accurately quantify SOD in various sediments exists. The lack of an integrated, universally applicable method for determining SOD necessitates the development of a CE-wide standard procedure to analyze SOD in riverine, lacustrine, and estuarine environments.

Initial investigations by the U.S. Army Engineer Waterways Experiment Station (WES) consisted of conducting a literature review and hosting a workshop to determine the state of the art of SOD research (Cerco, Gunnison, and Price 1992). The WES is currently conducting laboratory investigations interactively with model development to accurately measure, evaluate, and predict SOD for CE water resource projects. This paper describes initial laboratory investigations conducted at the WES to determine some SOD-regulated processes of DO depletion and nutrient release from a lacustrine sediment.

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Materials and Methods

A ponar grab dredge was used to collect three 5-gal buckets of sediment from Brown's Lake, WES, Vicksburg, MS. The buckets were sealed and refrigerated at 4 °C. Brown's Lake sediment, a fine-textured sediment composed predominantly of clay, was selected because of its availability, known physical characteristics, and use in past studies.

Test tube studies

Study I. Test tubes simulating a bedded sediment with an overlying water column were used to determine nutrient release and DO consumption of the sediment. Fifty-milliliter glass test tubes were loaded with 10 g of sediment amended with 1 percent organic matter. Freeze-dried *Hydrilla* was used for all organic matter amendments. The sediment was covered with 40 ml distilled-deionized water, and the tubes were sealed with rubber stoppers. All tubes were incubated at room temperature (23 °C). Sampling for DO, ammonium-nitrogen, and orthophosphate-phosphorus was conducted over a 36-day period. Samples were taken on days 0, 1, 3, 7, 11, 31, and 36. All samples were taken in triplicate, and each vessel was sacrificed at the time of sampling. Ammonium-nitrogen was determined using the phenate standard method 4500-NH_3 D (Standard Methods, 17th ed., 1989). Orthophosphate-phosphorus was measured using the ascorbic acid standard method 4500-P E (Standard Methods, 17th ed., 1989). A Shimadzu model UV-160A spectrophotometer (Shimadzu Scientific Instruments, Inc., Houston, TX) was used to measure final concentrations. Dissolved oxygen was measured with an Orion model 880 DO/BOD meter (Orion, Boston, MA).

Study II. Tests in study II were treated identically to study I, except samples were also taken for nitrate-nitrogen and total organic carbon (TOC). Determination of nitrate-nitrogen was carried out using the cadmium reduction standard method 4500-NO_3^- E (Standard Methods, 17th ed., 1989). TOC concentrations were determined with either a model 915A Beckman Total Organic Carbon Analyzer (Beckman, Inc., Irvine, CA) or a Shimadzu model 5050 Total Organic Carbon Analyzer (Shimadzu Scientific Instruments, Inc.). Two additional sets of tubes were used, having sediment amended with 5 and 10 percent organic matter, respectively. Sample times also varied; samples were taken on days 0, 1, 2, 6, 11, 16, 23, and 36.

Column study

Large Plexiglas columns were used to determine SOD and the release of nutrients from a bedded lacustrine sediment to the overlying water column (Figure 1). The large volume of water used in this study (15.7 L) allowed for sufficient sample to analyze for parameters without having to replace water lost through sampling. Columns were loaded with 15 cm of sediment (3,600 g) and overlain with 102 cm water (15.7 L). A 2-cm layer of mineral oil was used to seal the system from atmospheric contact. A plunger was lowered into the water column of each unit and left in place (Figure 1). The columns were incubated at 25 °C in an environmental chamber. The plungers were used to thoroughly mix the water columns daily. Dissolved oxygen, ammonium-nitrogen, orthophosphate-phosphorus, nitrate-nitrogen, and TOC were measured over a period of 3 months. Four samples were taken the first week, on days 0, 1, 5, and 8. All other samples were taken at 7-day intervals starting from day 8. All sampling methods were identical to those in the test tube studies.

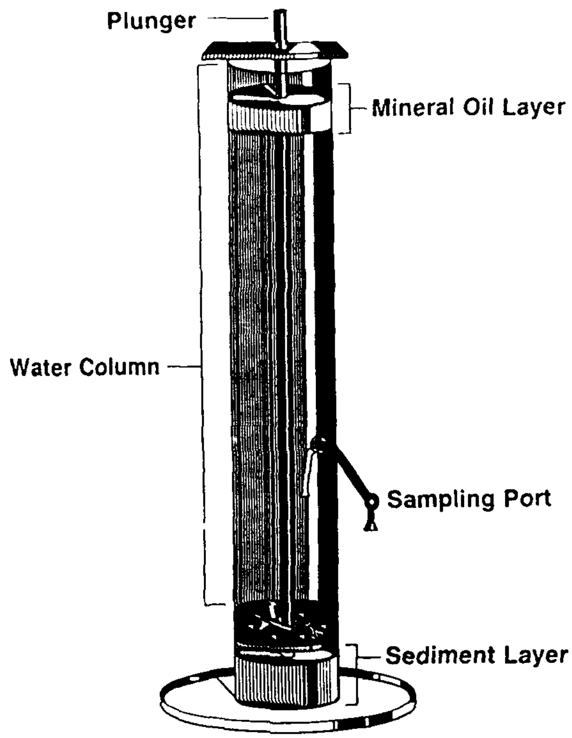

Figure 1. Plexiglas column used to determine SOD and nutrient release from Brown's Lake sediment

Results

Test tube study I

Dissolved oxygen concentrations dropped from 9.0 to 0.20 mg/L in 36 days. Initially, DO rapidly declined; it then leveled off, reaching a steady state in 1 week (Figure 2). Sediment oxygen demand, calculated from DO concentrations, is described as sediment flux as follows:

$$SOD(mg/m^2/day) = \frac{C_2 - C_1}{t} \cdot \frac{V}{A}$$

where

C_2 = concentration at end of sample interval
C_1 = concentration at start of sample interval
t = time of interval
V = volume of water in the test
A = cross-sectional area of sample vessel

The SOD flux ranged from -300 to -3.0 mg/m^2/day (Figure 2). The negative numbers indicate DO movement into the sediment. Positive numbers would represent release from the sediment.

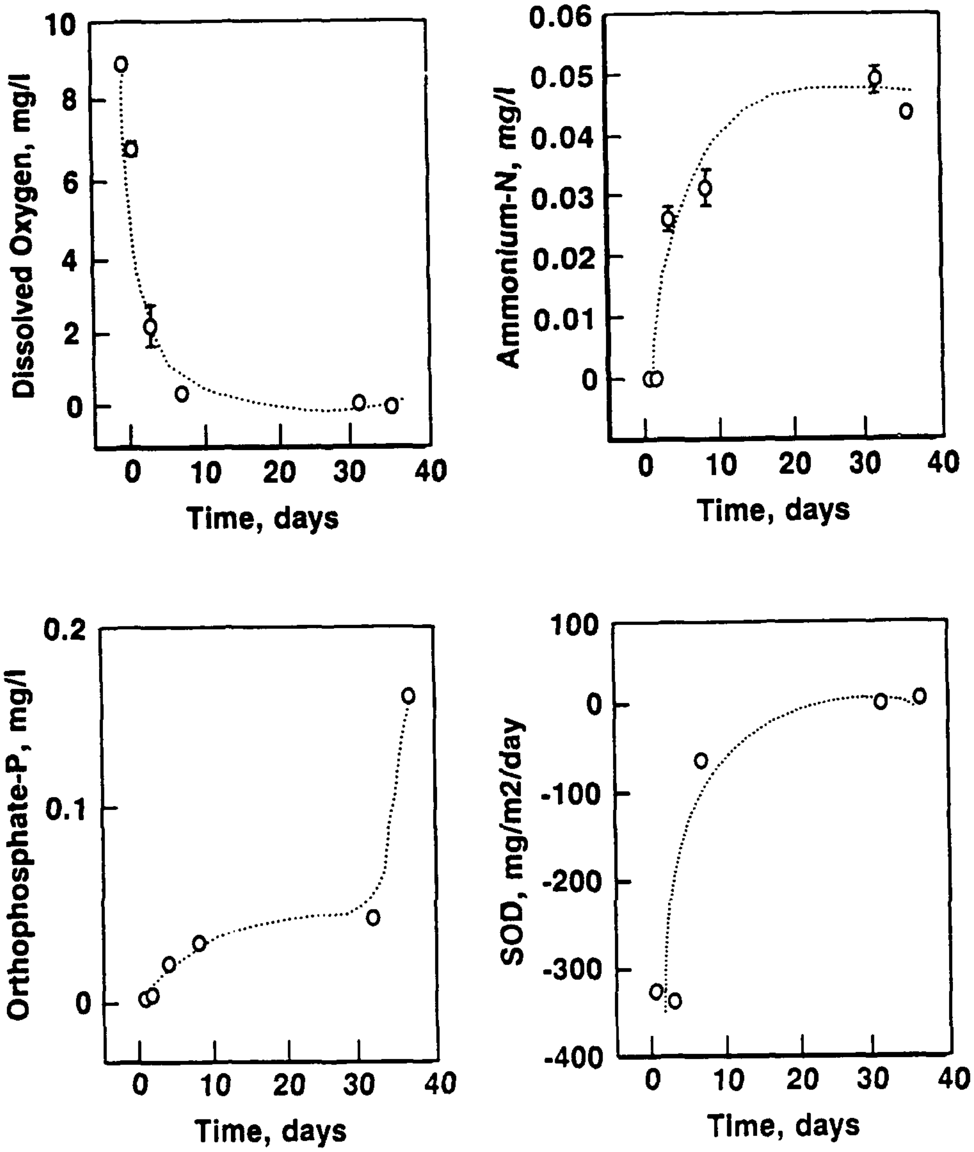

Figure 2. Dissolved oxygen and nutrient concentrations measured in overlying water

Ammonium-nitrogen levels measured in the overlying water increased over time reaching a peak of 0.048 mg/L at 30 days and leveling off to 0.042 mg/L (Figure 2). Orthophosphate-phosphorus concentrations also increased, measuring 0.165 mg/L in 36 days. A sharp increase in the concentration of this constituent occurred between sample days 31 and 36 (Figure 2).

Test tube study II

Dissolved oxygen concentrations ranged from 8.80 mg/L to an average of 0.29 mg/L during the 36-day test interval (Figure 3). The tubes amended with 10 percent organic matter showed the lowest DO concentration of 0.19 mg/L by day 36; but overall, DO levels of the three organic matter treatments did not differ significantly. Sediment oxygen demand ranged

from -300 mg/m^2/day on day 1 to -2.0 mg/m^2/day on day 36 (Figure 3). Ammonium-nitrogen concentrations increased in all three treatments over time (Figure 4). Initial concentrations in the overlying water measured 0.002 mg/L; final test concentrations on day 36 averaged 0.068 mg/L. Ammonium concentration data revealed no significant difference between the three treatments. Orthophosphate-phosphorus concentrations increased from 0.003 to 0.140 mg/L on sample day 6, and decreased to an average of 0.066 mg/L by day 36 (Figure 4). The different amendments showed no significant effect on phosphorus concentrations. Concentrations of nitrate-nitrogen initially increased from 0.023 mg/L to an average of 0.118 mg/L by sample day 2, and decreased to an average of 0.041 mg/L on sample day 36 (Figure 4). The three sample treatments showed no significant difference over time between the different amendment concentrations. Total organic carbon concentrations increased from 0 mg/L to an average of 32.8 mg/L by day 6 (Figure 4). Following this, TOC levels began decreasing to an average final concentration of 12.8 mg/L, with the 10 percent organic matter-amended tubes maintaining the highest TOC of 14.5 mg/L on sample day 36.

Column study

Dissolved oxygen concentrations decreased from 9.15 to 0.60 mg/L in 50 days (Figure 5). An initial rapid decrease in DO occurred up to day 7; this was followed by a slower decline. Sediment oxygen demand ranged from -2,000 to -2.0 mg/m^2/day by day 50 (Figure 5). Ammonium-nitrogen concentrations increased from 0.002 to 0.024 mg/L by day 7, then fell back to <0.01 mg/L (Figure 6). Orthophosphate-phosphorus increased to 0.45 mg/L on day 19 and began to decline, reaching 0.029 mg/L by day 50 (Figure 6). Nitrate-nitrogen rose to 0.228 mg/L on day 12, then decreased to 0.007 mg/L, and remained relatively stable for the remainder of the study period (Figure 6). Total organic carbon measurements fluctuated, reaching a peak of 11.7 mg/L by day 19 and decreasing to 1.77 mg/L by sample day 50 (Figure 6).

Discussion

The flux of oxygen-demanding substituents from bottom sediments into the overlying water column can give information about the potential sediment oxygen demand (Klapwijk and Snodgrass 1986). Both test tubes and 20-L columns were used to simulate bedded sediments with an overlying water column. The systems were allowed to develop anaerobic conditions naturally, and nutrient release and DO depletion were measured over time. The test tubes were initially used to allow for faster formation of anaerobic conditions, and to avoid the replacement of water removed through sampling. The second tube study was initiated to compare the SOD rates of sediments containing different concentrations of organic matter. Tubes were used to avoid water replacement, and to allow for a greater number of samples. The large water volume of the columns allowed water lost through sampling to be replaced without creating major changes in the water column chemical composition.

Test tube study I

Dissolved oxygen concentrations in this study showed a decrease in the rate of DO depletion as oxygen levels fell below 2 mg/L. Similar findings were reported by Wang (1981), who observed sharp initial decreases of residual oxygen followed by a tendency to taper off once levels fell below 2 mg/L.

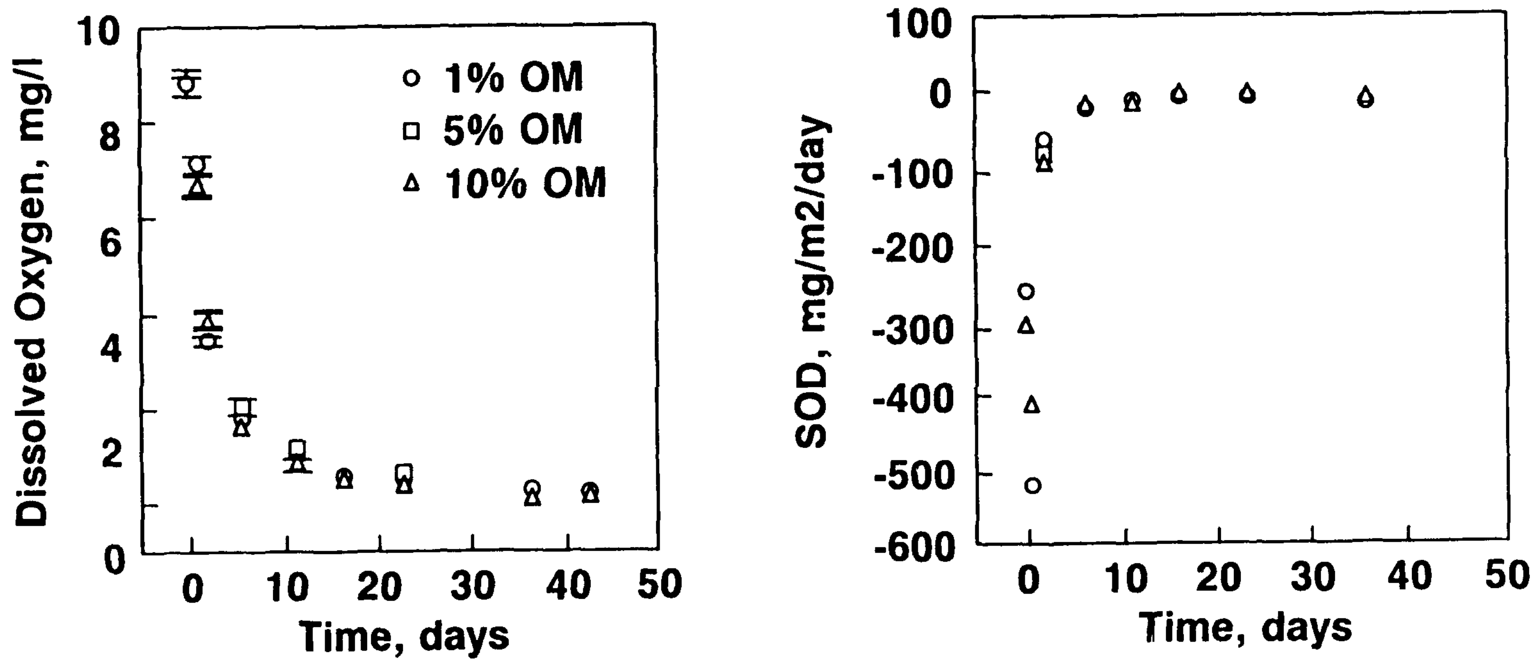

Figure 3. Dissolved oxygen and SOD flux measured in overlying water

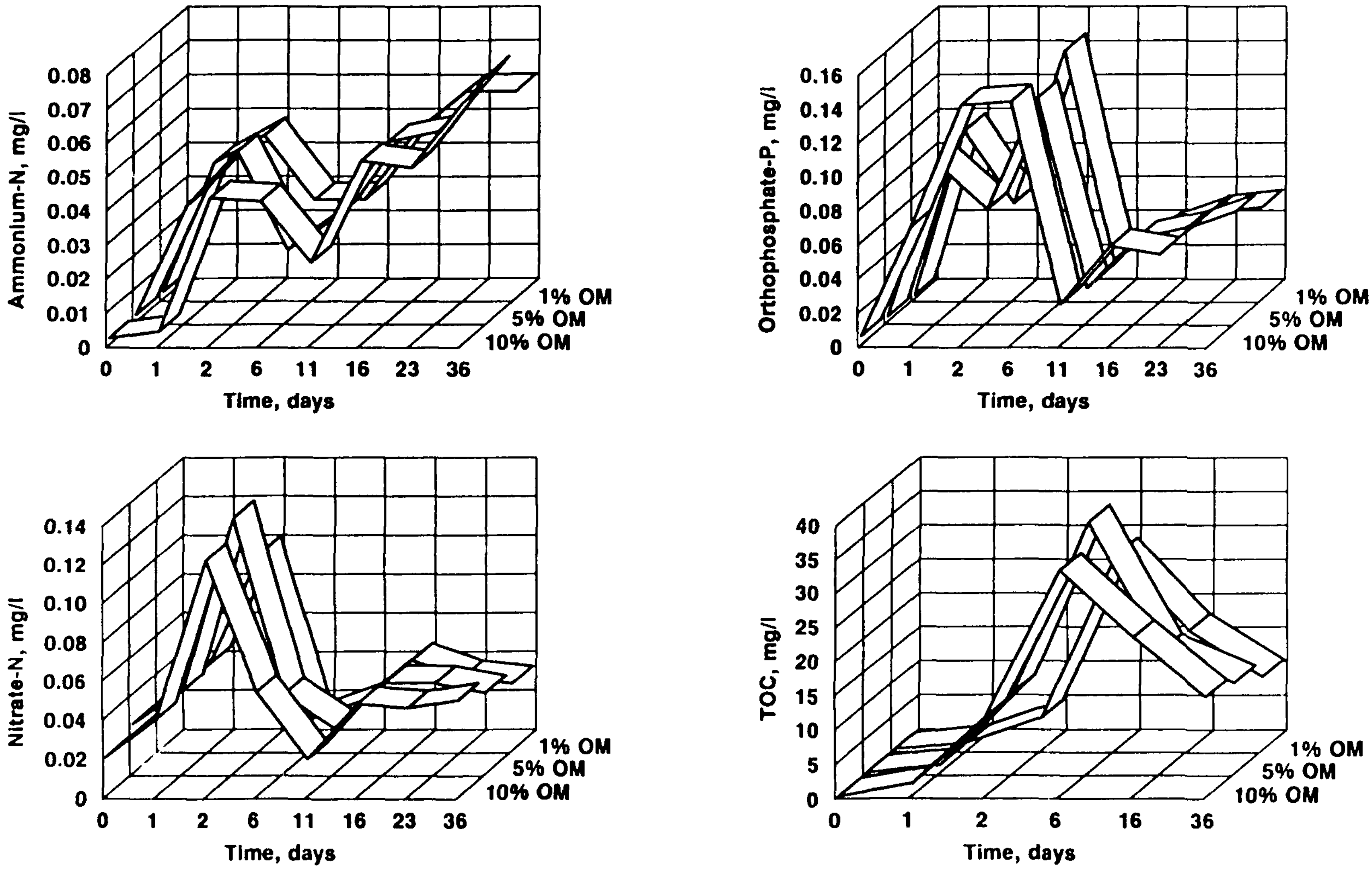

Figure 4. Nutrient and TOC concentrations released from Brown's Lake sediment

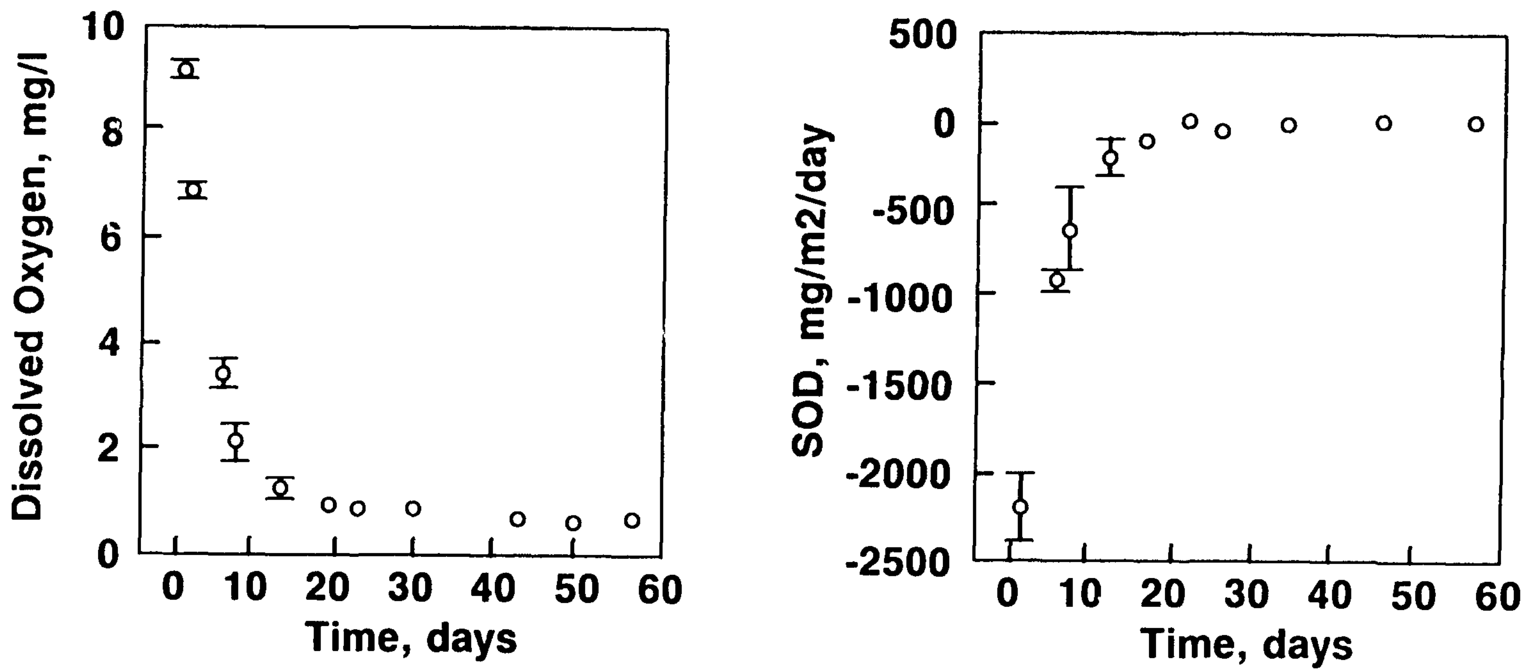

Figure 5. Dissolved oxygen measured and SOD flux calculated in column study

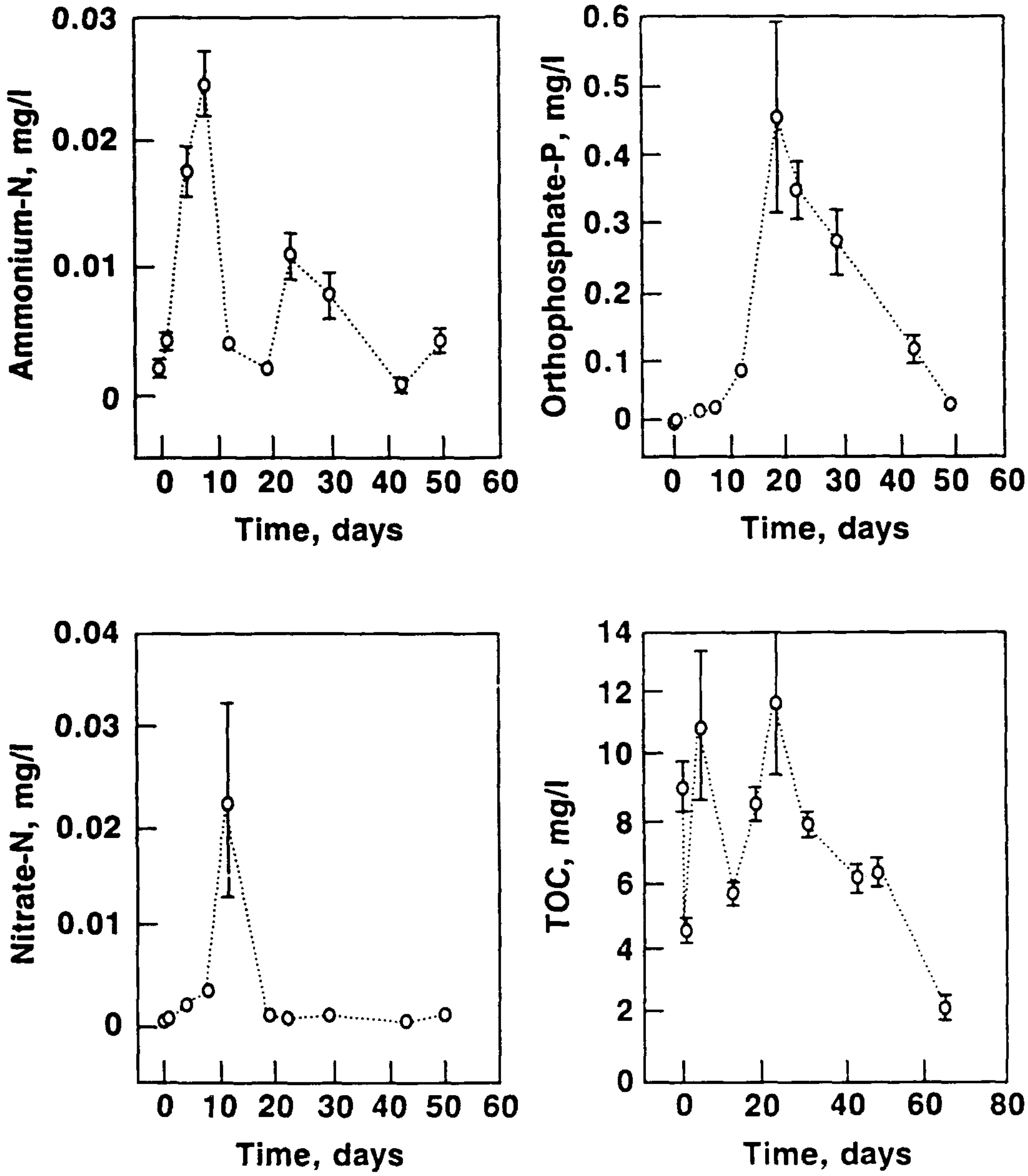

Figure 6. Nutrient and TOC concentrations released from Brown's Lake sediment in column study

Ammonium will diffuse to the surface of a sediment and, if oxygen is present, will undergo nitrification. Ammonium-nitrogen levels in the test increased over time and leveled off after day 15. If a system is anaerobic, nitrification does not occur; this is observed as an increase in ammonium in the water column (Klapwijk and Snodgrass 1986). Ammonium levels may have leveled off as the result of limitations in the organic nitrogen levels in the sediment.

In anaerobic systems, microbial activity may result in direct mobilization of inorganic phosphorus through the degradation of organic matter or through dissolution of

phosphate-adsorbing iron oxyhydroxides (Ryding 1985). The sharp increase in orthophosphate concentrations by day 36 of the sample interval may have resulted from these processes.

Test tube study II

No significant differences were observed in DO depletion or nutrient release among the three treatment levels. These results suggest that the rate of oxygen supply to the sediment limited the utilization of these substrates. Dissolved oxygen depletion rates exhibited the same behavior as in study I. The SOD rate mirrored the DO depletion rate. Dissolved oxygen concentrations in water overlying sediments can be considered a factor affecting SOD rates; as the DO concentration decreases, so does the SOD rate (Hicks 1992).

Ammonium-nitrogen levels increased in all amendments, but did not level off as in study I. This may be a result of the addition of a source of organic matter, which allowed for a prolonged period of mineralization of organic-nitrogen to ammonium. Differences among treatments were not significant.

Phosphorus release can be promoted in anaerobic bottom waters as the result of the chemical reduction of Fe^{3+} to Fe^{2+}, and dissolution of the phosphate-adsorbing ferric oxyhydroxide (Ryding 1985). Orthophosphate-phosphorus concentrations initially increased and then decreased in all three amendments. The decline in phosphorus concentrations may be a result of phosphate interacting with Al^{3+} or Ca^{2+}. Interactions between phosphorus and these cations can result in the formation of fairly insoluble precipitates, or adsorption of PO_4^{3-} to colloidal oxides, hydroxides, and carbonates (Ryding 1985).

Nitrate-nitrogen concentrations increased as initial releases of ammonium were nitrified. Nitrate levels decreased markedly when DO concentrations reached approximately 1 mg/L. Nitrate did not completely disappear from the system, indicating the presence of some oxygen in the overlying water. Nitrate will move downward by diffusion and undergo denitrification in the sediment (Ponnamperuma 1972). However, as stated by Gunnison, Chen, and Brannon (1983), very little denitrification takes place until all oxgyen has been depleted.

Total organic carbon measured in the overlying water began to decrease when DO levels fell below 1 mg/L. Gunnison, Chen, and Brannon (1983) reported that a decrease in concentration of soluble TOC in the water column was strongly correlated with a decrease in DO depletion rate. Dissolved oxygen depletion rates decreased by day 11 of sampling, corresponding to a decrease in TOC in the same period.

Column study

Dissolved oxygen concentrations decreased in the columns, but only to an average of 0.60 mg/L. However, DO levels did exhibit the same behavior as in the test tube studies. Levels of DO may have been prevented from falling lower by oxygen diffusion through the mineral oil seal on the columns. The SOD rate was insufficient to overcome this influx of DO.

Almost all of the mineralizable nitrogen in sediments is converted to ammonium within 2 weeks of flooding (Ponnamperuma 1972). Ammonium-nitrogen exhibited an initial increase, but then fell and leveled off as in test tube study I. This may have resulted from limited organic nitrogen levels in the sediment. However, the nitrate data suggest that

substantial levels of ammonium may have been converted to nitrate up to day 15. Following this, nitrate production ceased, and denitrification removed most of the nitrate.

Orthophosphate exhibited the same behavior as in the second test tube study, with an initial increase, followed by a decline. The phosphate may have become tied up as aluminum and/or calcium phosphate precipitates.

Nitrate-nitrogen levels initially increased in the water columns, but decreased when DO levels reached approximately 1.5 mg/L. When nitrate is formed in the aerobic layer, it can diffuse both up into the water column and down into the anaerobic soil layer; however, net diffusion is downward into the sediment, where the nitrate undergoes denitrification (Patrick 1990). The larger volume of sediment in the column study may have allowed for a strong enough SOD to deplete the nitrogen levels in the overlying water.

Initial fluctuations in TOC concentrations may have been a result of sample treatment. Initial samples were frozen as a result of equipment failure. TOC levels followed the same behavior as in test tube study II; that is, as the DO depletion rate decreased, so did TOC concentrations.

Conclusion

All three studies confirmed that DO levels are directly related to SOD processes. SOD has a major effect upon the amount and rate of release of key nutrients from a lacustrine sediment.

The test tube and column studies determined that ammonium, orthophosphate, and nitrate releases are directly related to DO depletion. TOC concentrations in both studies exhibited similar behavior by decreasing as DO depletion rates decreased. These results indicate that DO demand was strongly related to soluble TOC.

These initial studies revealed the need for further investigations of sediment-water interactions and SOD-regulated processes. Sediments exhibiting a broad range in characteristics should be evaluated to better understand the effects of SOD on nutrient release in different aquatic environments.

Acknowledgments

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Application and Use of RECOVERY

by
Carlos E. Ruiz[1]

Abstract

RECOVERY is a PC-based screening model to assess the impact of contaminated bottom sediments on surface waters. The analysis is limited to organic contaminants and to surface waters that are well mixed. The contaminant is assumed to follow linear equilibrium sorption and first-order decay kinetics.

The physical representation of a system by RECOVERY consists of a well-mixed surface-water layer underlain by a vertically stratified sediment column. The sediment is well mixed horizontally but segmented vertically into a well-mixed surface (active) layer and deep sediment. The deep sediment is segmented into contaminated and clean sediments regions. RECOVERY is designed for interactive implementation via a personal computer. The program allows the user to rapidly generate and analyze recovery scenarios for contaminated sediments.

The RECOVERY model was applied to a contaminated quarry for confirmation purposes. The results were compared to temporal data collected in 1972 and a sample collected 5 years later. The model was also compared to a verified fate and transport model. Overall, RECOVERY shows potential as a screening model for assessing the exposure of organic contaminants originating from sediments in aquatic systems. Example applications of RECOVERY will be demonstrated. A test case with capping contaminated sediments, a no-capping alternative, and the effects of volatilization will also be presented.

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Variations of Oxygenation Efficiency: Concepts and Performance in Richard B. Russell

by
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Abstract

Large-scale in-lake oxygenation has continued in Richard B. Russell Lake (RBR) since 1985 when RBR Dam began hydroelectric operation. However, the effect this system has had on lake dissolved oxygen distributions has varied with its operation. The oxygenation system is composed of two spatially separated arrays of diffusers, one of which, the pulse system, has been employed more often for achieving release concentrations of at least 6 mg/L dissolved oxygen. A continuous system located approximately 1 km upstream has also been employed to help achieve this goal.

Accidental interruption of system operation supplied the first "experimental" observation of its effect on the lake and release waters from RBR dam. The absence of the system resulted in a decrease of approximately 2 mg/L in the RBR release, indicating that that was the system's contribution to the dissolved oxygen concentration. Off-gas analysis of bubbles emerging from the lake surface during normal operation showed great variation of oxygen content (20 to 80 percent oxygen released to the atmosphere).

In-lake studies under way will further assess the effect that the oxygenation system has on the distribution of oxygen, as well as on distributions of reactive chemicals such as dissolved metals. These studies may also indicate how off-gas analyses may be used as a quick method of assessing the approximate efficiency of operation of individual or small groups of diffusers, and how this might be used in the field as an indication of the location of possible diffuser malfunctions.

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SUMMARY OF WORKSHOPS

The Water Operations Technical Support and Dredging Operations Technical Support programs sponsored two workshops in conjunction with the 9th Seminar on Water Quality. The workshop "Use of Water Quality Models" was attended by 20 participants. This workshop introduced one- and two-dimensional water quality modeling using CE-QUAL-RIV1 and CE-QUAL-W2, respectively. The workshop "Sediment-Water Interactions" was attended by 35 participants. This workshop provide the latest information on techniques for evaluating sediment-water interactions with respect to the fate of nutrients and contaminants.

Workshop participants were asked to complete an evaluation form. Each participant was asked to rate the workshops in several categories and to provide comments. Below is a summary of the rating evaluations for each workshop.

Use of Water Quality Models Workshop

Category	Excellent	Good	Fair	Poor
Room	2	6	0	0
Visual aids	3	5	0	0
Presentations	3	5	0	0
Relevance	4	3	1	0
Objectives met	3	4	1	0
Adequate coverage of topics	5	2	1	0
Instructor knowledge	7	1	0	0

Sediment-Water Interactions Workshop

Category	Excellent	Good	Fair	Poor
Room	7	10	0	0
Visual aids	4	13	0	0
Presentations	5	11	1	0
Relevance	4	9	3	1
Objectives met	3	12	2	0
Adequate coverage of topics	4	11	2	0
Instructor knowledge	13	4	0	0

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